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BIODIVERSITY IN HARVESTED AREAS IN RELATION TO THE DISPERSION OF STANDING LIVE TREES AND SNAGS:

1998/99 YEAR-END REPORT

Biodiversity In Harvested Areas In Relation To The Dispersion Of Standing Live Trees And Snags: 1998/99 Year – End Report

By

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1999

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DISCLAIMER

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EXECUTIVE SUMMARY

In this project, we are evaluating the extent to which biodiversity in harvested areas is affected by the size, number, and spacing of tree clumps that are left after harvest. Twelve experimental cut-blocks were created in each of two study areas. One experimental area is in pine/spruce-dominated forests near Whitecourt, Alberta, and the other is in aspen-dominated forests near Grande Prairie, Alberta. Within each of the cut-blocks in the experimental areas, 3% of the pre-harvest merchantable trees were retained in clumps. One third of the experimental cut-blocks had clumps with a mean of 0.64 ha (1.58 acres), one third had clumps with a mean of 0.14 ha (0.35acres), and one third had clumps with a mean of 0.04 ha (0.10 acres). This research complements on-going research at the Alberta Research Council, the University of Alberta, the Network of Centers of Excellence, and forest industries throughout the Alberta by evaluating how residual clump size and spacing affects biota rather than evaluating how differences in amount of residual materials affects biota. All residual tree clumps evaluated in this study can be implemented by forest companies that leave some canopy trees in their harvest blocks. This project started April 1997 at which time biota were surveyed in the treatment areas prior to harvest. During the winter 1997/98 the experimental cut-blocks were harvested. Some of the biota in the post-harvest cut-blocks were re-surveyed during 1998 with the remainder to be surveyed during 1999. Although the primary focus of this project is on the biotic community, we also evaluate the fall-down rates of trees and snags that are left standing in the experimental areas.

Within both study areas we categorized the experimental stands into four groups to reduce variance and increase statistical power of our tests. Composition and density of trees and snags, volume of downed woody material, bird communities, and bat abundance varied among the stand groups pre-harvest. After accounting for differences among stand groups, however, forest structure and biotic communities did not vary among treatments. Thus, differences we found among treatments after harvest can be attributed directly to variation in size, number and spacing of tree clumps retained at harvest.

During 1998 we conducted comprehensive surveys of the experimental cut-blocks for trees, snags, downed woody material, bats, beetles, and conducted cursory surveys for birds. Fall down rates of trees were low in both study areas (< 15% and 5% in the Whitecourt and Grande Prairie study areas respectively) but significantly lower in the large clumps than in the smaller clumps. More downed woody material was present in cutblocks with small clumps than in cutblocks with large clumps. For both beetles and birds, the communities in large clumps were more similar to those in unharvested forest than were the communities in small clumps. Bat abundance, however, did not differ among treatment types. Patterns were consistent for both the Whitecourt and Grande Prairie study areas, and suggest that cutblocks with large clumps retain more characteristics of unharvested forest than do cutblocks with small clumps. Not all taxonomic groups, however, were surveyed during 1998.

During 1999 we will conduct comprehensive surveys of the experimental cut-blocks for bats, herbs, shrubs, and birds. We will prepare a final report incorporating all data by March 2000. In addition, field tours will be conducted in the experimental cut-blocks, and verbal presentations will be given to all funding partners of the project by March 2000. The final report will include a summary chapter describing similarities and differences among biotic groups that occurred as a consequence of retaining trees in many small clumps versus few large clumps at harvest.

Deliverables

As part of developing and conducting this research we have worked closely with government regulatory agencies and the forest industries to ensure that the results will be applicable to these organizations. We will continue to work with these organizations and present results (to groups and to individuals) as results become available. Specific deliverables include:

1) Mid-year and year-end progress reports.

2) Descriptions of how the spatial pattern of standing trees and snags (i.e., variation in size and spacing of tree clumps) affects the vegetation, arthropod, bird, and bat communities within harvested areas.

3) Descriptions of the fall-down rates for trees and snags that are retained both within clumps and between clumps within harvested areas.

4) Recommendations for clump size and clump spacing that are most suitable for meeting biodiversity goals.

5) Presentations of research findings in client reports, primary-journals, and at conferences.

6) Participation in workshops, planning seminars, and committees that assess and regulate alternative harvest practices in Alberta.

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CHAPTER 1. BACKGROUND AND RATIONALE

Jim Schieck

Background

As a consequence of having a thriving forest industry in Alberta, much of Alberta's boreal forest has been allocated to be harvested either for dimensional lumber or for pulp and paper. The economic benefits of this intensive forest harvesting are viewed positively by Alberta's population. The ecological ramifications of extensive forest harvest, however, are poorly understood and this lack of knowledge hampers management decisions and land use planning. Consequently, the forest industry, and government organizations, in Alberta have been conducting research to develop forest practices that are ecologically sound, economically feasible, and sustainable.

Short rotation practices will truncate forest succession over much of the landscape and, in some areas, may result in old forests becoming relatively scarce. In addition, the removal of most of the merchantable trees from harvested areas will alter natural floral and faunal communities in those areas and may affect biotic recovery. Finally, large live trees and snags provide suitable micro-habitats for many plants, and nesting and foraging habitats for many animals, but with conventional harvest practices, few large trees and snags will be present in harvested landscapes.

Management strategies are being developed to produce harvest patterns that are as similar as possible to natural disturbances. Since large trees and snags are abundant following natural disturbances, but are greatly reduced in harvested areas, forest managers have been recommending that as many trees and snags as is socially and economically feasible be retained within harvested areas. Clumps of standing trees and snags in harvested areas may function similar to skips in wildfire areas, and act as refugia for some species while the harvested forest re-grows. If residual standing live trees continue to grow, they will become large live trees and may produce some large snags in mid- and late-rotation forests. These large live trees and snags may provide some of the structures and micro-habitats that are normally found only in post-rotation aged forests.

Given the complexity of natural ecosystems, it is not surprising that the ecological and biodiversity impacts of these alternative harvest/silviculture practices are poorly understood. Only by experimenting with new techniques in the field, and monitoring the resulting ecosystems throughout a harvest rotation, will we begin to understand the potential impacts and benefits of each technique. To this end, research has been conducted by scientists within the Alberta Research Council, the Alberta Environmental Protection, the University of Alberta, the Network of Centers of Excellence, and forest industries throughout Alberta's boreal forests.

Preliminary results from past and on-going stand-level research indicates that biodiversity in harvested areas is positively related to the amount of standing trees and snags retained in those areas. However, only anecdotal information is available to evaluate whether the pattern in which those trees are retained also affects the ecology and biodiversity in harvest areas. The present study will fill that gap, by evaluating whether the biota present in harvest areas with many small clumps of trees differs from that present in harvest areas with few large clumps of trees. If clump size has a direct impact on biota, then by leaving trees and snags in the appropriate clump sizes, forest managers may obtain benefits to biodiversity at little additional cost.

The proposed experimental treatments will be monitored throughout a harvest rotation (extended monitoring is not part of the present proposal) to enable researchers to evaluate changes in trees, understory vegetation and wildlife over time. This controlled experimental study will contribute to the adaptive management of boreal forests and will provide information that forest managers can use when designing harvest plans.

Scientific Rationale

All else being equal, it would be beneficial to make harvest areas as similar as possible to natural disturbances since native biota evolved in an ecosystem with natural disturbances (Urban et al. 1987). The dominant natural disturbance in the boreal forests of North America is wildfires (Eberhart and Woodard 1987, Hunter 1993), and cut-block size and shape can be patterned after wildfires. There are limitations, however, to matching the amount of wood remaining post-harvest to that post-wildfire because only a small percentage of the wood is combusted during a wildfire but most of the wood is removed during harvest (Lee et al. 1997). Given this discrepancy, it is necessary to manage the relatively few trees and snags that are retained in harvest areas to achieve the greatest benefit for biota.

Following wildfires, there are groups of unburned trees and snags (fire skips) embedded within the burned areas (Eberhart and Woodard 1987). These fire skips could be emulated in harvest scenarios by leaving clumps of live trees and snags within cut-blocks. Fire skips and residual clumps of trees probably retain some of the structures and micro-habitats that were present in the forest prior to disturbance and may act as refugia for some native species. Small clumps may be sufficient for a few species, but larger clumps probably act as refugia for more species (Saunders et al. 1991). For example, some animals require large undisturbed areas for protection and foraging (Porneluzi et al. 1993), and some plants only exist in micro-habitats found in large undisturbed areas (Brothers and Spingarn 1992).

The refugia potential of residual clumps of trees is probably most important during the first few decades following harvest because many boreal forest species live at low densities in young forests (Schieck et al. 1995, Kirk et al. 1996). Biota that are present in the residual clumps may disperse from these clumps into harvest areas once the new cohort of trees is 20 to 50 years old. Distance between clumps, and species dispersal abilities, will affect the rate and dynamics of biotic recolonization in harvested areas (Saunders et al. 1991, deMaynadier and Hunter 1995). The degree to which tree clumps act as refugia in harvested areas, and the appropriate dispersion of those clumps to enhance dispersal of biota, have not been evaluated in boreal forests.

In addition to the refugia potential of large tree clumps, there may be management value to retaining small clumps, or single live trees, scattered throughout harvest areas. In areas where intensive forest harvest will decrease the amount of old forest in the landscape, it may be beneficial to produce old-forest structures and micro-habitats in mid seral stages (Spies et al. 1991). Many of these old-forest structures and micro-habitats are associated with large live and dead trees (Hansen et al. 1991, Schieck et al. 1995). Consequently, if some merchantable trees are retained as standing live trees at harvest, and these trees do not get blown down, they will become large by mid-rotation and may accelerate biotic recovery in the harvest areas. With this management strategy, single trees or many small clumps of trees, scattered over the harvest area, may have greater impact on biota than if all standing trees and snags had been retained in few large clumps because old-forest structures and micro-habitats would then be dispersed throughout the harvest area.

The most appropriate placement of standing trees and snags differs between "managing for refugia" and "managing for the production of dispersed large trees". Both patterns may be useful, but only by experimentally manipulating size of clumps and distance between clumps, and then studying the biota living in the areas throughout a rotation will we understand the biological and management implications of each scenario.

In this study, we established three treatments with different clump sizes and distances between clumps in two experimentally harvested areas to evaluate whether biota differ among treatments. In the present 3-year study we will evaluate differences among treatments that are present immediately following harvest. The experimental areas will be maintained for the rest of the harvest rotation however, and we will evaluate differences among treatments periodically throughout succession (i.e., once the new cohorts of trees are well established, at mid-rotation, and just prior to the subsequent harvest).

Objectives

In the present study, we are evaluating how clump size and distance between clumps of residual live trees and snags affects biodiversity in harvested areas. All treatment areas were sampled before and after the experimental harvest. During 1997/98 we compared pre-harvest biotic communities among treatments to ensure that treatments were similar prior to harvest. Part of the post-harvest biotic communities were surveyed during 1998 and the remainder will be surveyed during 1999. We compared post-harvest biotic communities among treatments to evaluate whether biotic communities present within the first two years post-harvest were related to clump size and distance between clumps. In addition, we discussed similarities and differences of the present study to previous studies of post-harvest biota.

In this study we evaluated: (1) whether the bird community that has specialized foraging and nesting requirements for large live and dead trees varied among the three silviculture treatments, (2) whether the resident or migrant bird communities varied among treatments, (3) whether the bat community varied among treatments, (4) whether the carabid beetle community varied among treatments, (5) whether the herb community varied among treatments, (6) whether the shrub community varied among treatments, and (7) whether fall-down rates of standing trees and snags varied among treatments.

Literature cited

Brothers, T.S., and Spingarn, A. 1992. Forest fragmentation and alien plant invasion of central Indiana old-growth forest. Conservation Biology 6: 91-100.

deMaynadier, P.G., and Hunter, M.L. 1995. The relationship between forest management and amphibian ecology: a review of the North American literature. Environmental Review 3: 230-261.

Eberhart, K.E., and Woodard, P.M. 1987. Distribution of residual vegetation associated with large fires in Alberta. Canadian Journal of Forest Research 17: 1207-1212.

Hansen, A.J., Spies, T.A., Swanson, F.J., and Ohmann, J.L. 1991. Conserving biodiversity in managed forests. BioScience 41: 382-392.

Hunter, M.L. 1993. Natural fire regimes as spatial models for managing boreal forests. Biological Conservation 65: 115-120.

Kirk, D.A., Diamond, A.W., Hobson, K.A., and Smith, A.R. 1996. Breeding bird communities of the western and northern Canadian boreal forest: relationship to forest type. Canadian Journal of Zoology 74: 1749-1770.

Lee, P.C., Crites, S., Nietfeld, M., Van Nguyen, H., and Stelfox, J.B. 1997. Characteristics and origins of deadwood material in aspen-dominated boreal forests. Ecological Applications 7: 691-701.

Porneluzi, P., Bednarz, J.C., Goodrich, L.J., Zawada, N., and Hoover, J. 1993. Reproductive performance of territorial ovenbirds occupying forest fragments and a contiguous forest in Pennsylvania. Conservation Biology 7: 618-622.

Saunders, D.A., Hobbs, R.J., and Margules, C.R. 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology 5: 18-32.

Schieck, J., Nietfeld, M., and Stelfox, J.B. 1995. Differences in bird species richness and abundance among three successional stages of aspen dominated boreal forests. Canadian Journal of Zoology 73: 1417-1431.

Spies, T.A., Tappeiner, J., Pojar, J., and Coates, D. 1991. Trends in ecosystem management at the stand level. *In* Transactions of the fifty-sixth North American wildlife natural resources conference. *Edited by* R.E. McCabe, Wildlife Management Institute, Washington DC. pp. 628-639.

Urban, D.L., O'Neill, R.V., and Shugart, H.H. 1987. A hierarchical perspective can help scientists understand spatial patterns. Landscape Ecology 37: 119-127.

CHAPTER 2. STUDY AREA AND METHODS OVERVIEW

Jim Schieck and Christine Gray

This study is being conducted in two forest types in the boreal forest of western Alberta. The study area southwest of Whitecourt, AB (54° 10' N, 116° 20' W) was dominated by lodgepole pine (*Pinus contorta*) and black spruce (*Picea mariana*) that was approximately 105 years old and had approximately 1450 canopy trees per ha (590 canopy trees per acre). Throughout the area there is localized topographical relief resulting in pockets (<5% of the total area) of aspen (*Populus tremuloides*) and white birch (*Betula papyrifera*). Twelve stands that ranged in size from 27 to 46 ha, mean 34 ha (67 to 114 acres, mean 84 acres) were established in the study area (Fig. 2.1). These stands were categorized into four groups of three, based primarily on the type of vegetation that surrounded them and secondarily on the tree species composition in the area prior to harvest. Within each stand group one stand was randomly assigned to each of the three treatments (Fig. 2.1).

Stands were harvested between September 1997 and February 1998. Three percent of the pre-harvest volume was left in tree clumps and as many of the non-merchantable trees and snags as was possible were retained between clumps. Clumps were not placed on roads or on cut-lines, but otherwise they were dispersed in a regular pattern. To reduce wind-throw, clumps were circular and all trees within 3 m (10 ft.) of the perimeter of each clump were topped at 4-5 m (13-16 ft.) to create stubs. We have assumed that each stub contained one third of the volume of the original tree and have added sufficient area to each clump to replace the volume that was removed from the stubs. The number of clumps in a stand was determined based on the treatment being created and the size of the stand. Based on a stand area of 30 ha (74 acres), treatment one would have two large clumps with a diameter of 79.6 m (87.0 yd.), treatment two would have eight medium sized clumps with a diameter of 41.7 m (45.6 yd.), and treatment three would have 32 small clumps with a diameter of 22.7 m (24.8 yd). The actual area of the clumps was 0.43 - 0.61 ha (1.1 – 1.5 acre) for the large clumps, 0.13 – 0.14 ha (0.32 - 0.35 acre)for the medium clumps and 0.04 ha (0.10 acre)for the small clumps (see Appendix I for number of clumps in each stand). The harvested areas within the stands will be re-planted with 2-year old pine seedlings during the summer of 1999 and sprayed with herbicide during September 1999.

The study area southwest of Grande Prairie, AB (54° 50' N, 119° 20' W) was dominated by aspen trees that originated after a fire approximately 100 years ago. This study area has a few small depressions creating pockets of white spruce (*Picea glauca*), balsam poplar (*Populus balsamifera*), white birch (*Betula papyrifera*), willow (*Salix spp*.) and black spruce (*Picea mariana*). Within this study area there are approximately 750 canopy trees per ha (300 canopy trees per acre). Twelve experimental stands were created by harvesting blocks in a checker-board pattern such that approximately half of the study area was harvested and half of the area unharvested (Fig. 2.2). Stands ranged in size from 35 to 66 ha, mean 52 ha (86 to 163 acres, mean 128 acres). These stands were categorized into four groups of three, based primarily on the type of vegetation that surrounded them and secondarily on the tree species composition in the area prior to harvest. Within each group, one stand was randomly assigned to each of the three treatments (Fig. 2.2).

Stands were harvested between September 1997 and February 1998. Three percent of the pre-harvest volume was left in tree clumps and as many of the non-merchantable trees and snags as was possible were retained between clumps. Clumps were not placed on roads or cut-lines, but otherwise were dispersed in a regular pattern throughout the stand (Fig. 2.2). The number of clumps in a stand was determined based on the treatment being created and the size of the stand. Based on a treatment area of 50 ha (124 acres), treatment one would have two large clumps with a diameter of 97.7 m (107 yd.), treatment two

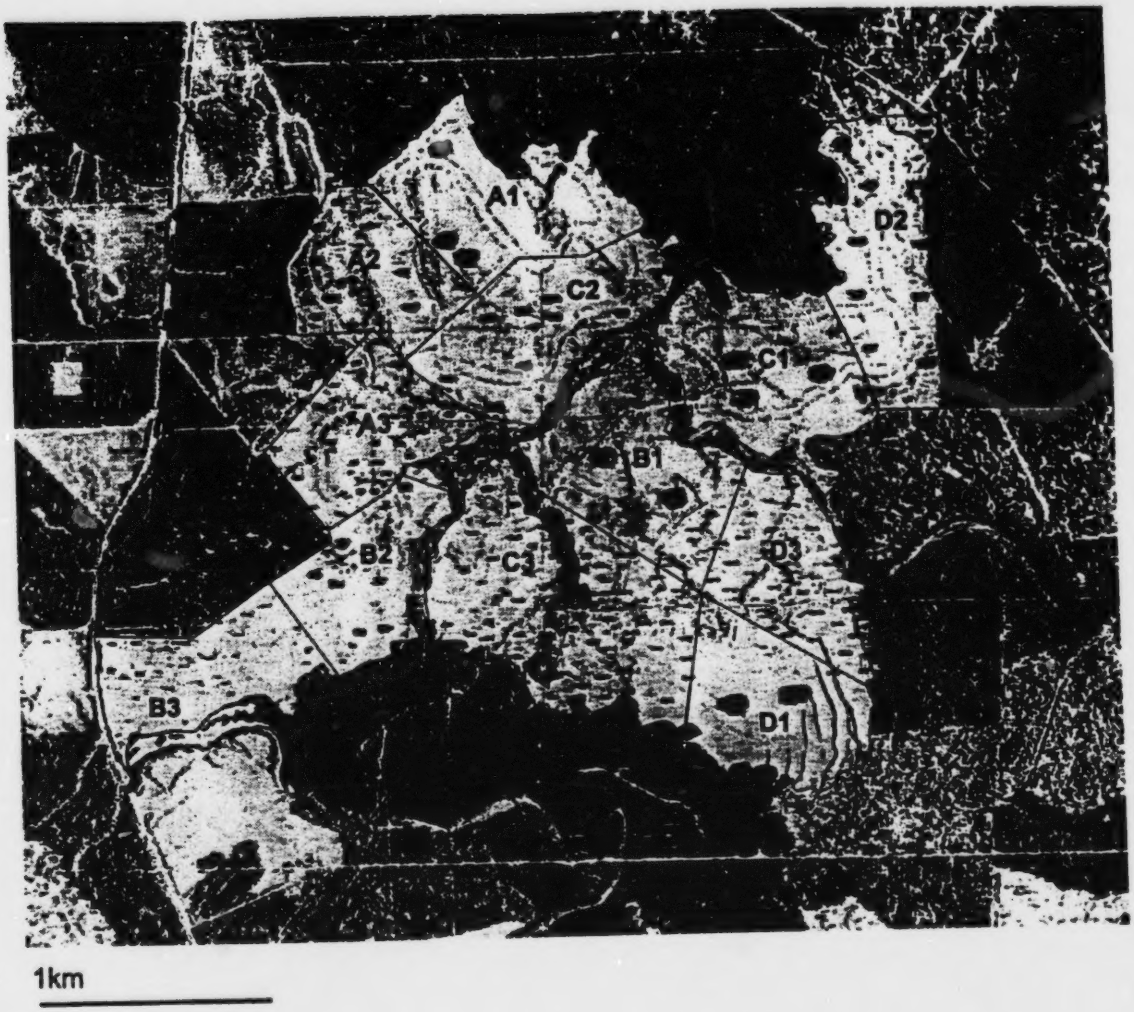
A1
D2
A2
C2
C1
A3
B1
B2
C3
D3
B3
D1
1km

FIGURE 2.1. Aerial photo showing the locations of stands and tree clumps in the Whitecourt study area.

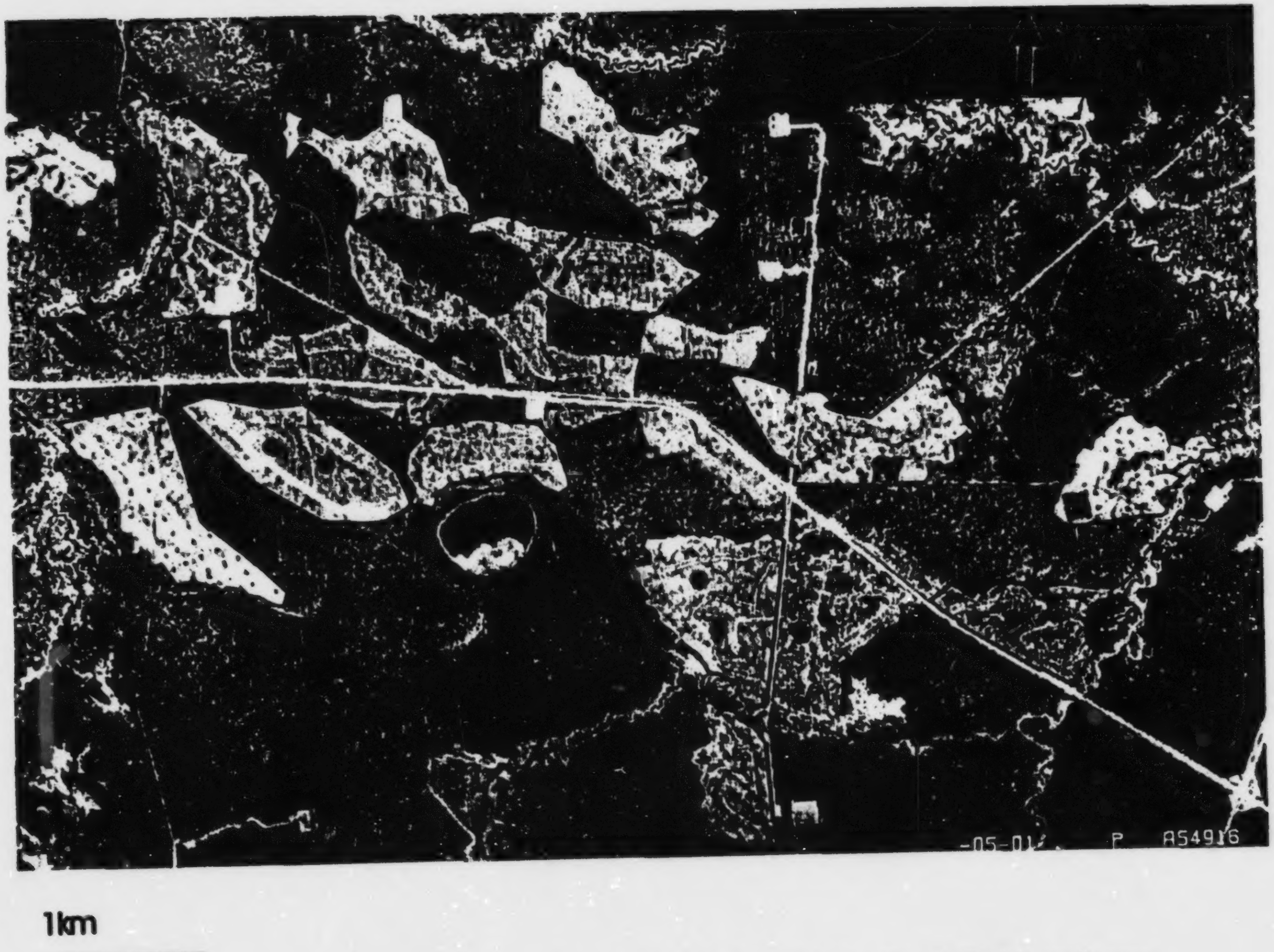

FIGURE 2.2. Aerial photo showing the location of stands and tree clumps in the Grande Prairie study area.

would have ten medium sized clumps with a diameter of 43.7 m (47.8 yd.), and treatment three would have 50 small clumps with a diameter of 19.5 m (21.3 yd.). The actual area is 0.57 – 0.88 ha (1.4 – 2.2 acre) for the large clumps, 0.15 ha (0.37 acre) for the medium clumps and 0.03 ha (0.07 acre) for the small clumps (see Appendix IV for the actual number of clumps in each stand). No post-harvest silviculture is planned for the stands.

To ensure that harvest patterns were consistent among treatments, the boundaries of the clumps of trees and snags that were to be retained at harvest were marked with colored plastic ribbon prior to harvest. No machine activity occurred within the residual clumps of trees during harvest.

General Survey Methods

Pre-harvest surveys of biota were conducted during the spring and summer of 1997. Ten 1-ha (2.47 acre) sites were established in the center of each stand (Appendix II and V) and all surveys of forest structure and biota were conducted in those central sites. Trees and snags (Chapter 3), downed woody material (Chapter 4), and understory vegetation (Chapter 5) were surveyed in a series of nested plots and transects (Fig. 2.3). Breeding birds were surveyed using a combination of point counts and strip transects (Chapter 7). Bats were surveyed using automatic remote bat detectors (Chapter 8).

Post-harvest surveys of biota were conducted during the spring and summer of 1998 and will continue during 1999 (Table 2.1). Some of the surveys were conducted within a sample of residual tree clumps (Appendix I and IV), cutover areas, and adjacent uncut forest. Where possible, the same clumps/areas were chosen for forest structure measurements and biota surveys. Residual trees and snags were characterized, and fall-down rated determined within residual tree clumps, 1-ha (2.47 acre) sites within the cutover area (Fig. 2.4), and ¼-ha (0.62 acre) sites in adjacent unharvested stands (Appendix III and VI) (Chapter 3). Downed woody material was surveyed along 25 m transects within residual tree clumps and 1-ha sites (Chapter 4). Understory vegetation was not surveyed in 1998 (Chapter 5). Carabid beetles were surveyed using pitfall traps along transects radiating from the center of residual clumps into cut-over areas (Chapter 6). Cursory bird surveys were conducted using point counts in residual clumps and cut-over areas (Chapter 7). Bats were surveyed using automatic remote bat detectors in residual clumps and cut-over areas (Chapter 8).

During 1999 different biota will be surveyed than during 1998 (Table 2.1). The fall-down of trees and snags that were marked during 1998 will be reassesed (Chapter 3). Understory vegetation will be surveyed using a series of nested plots within each 1-ha site (Fig. 2.4, Chapter 5). Birds will be surveyed through each 1-ha site using strip transects (Fig. 2.4) and within residual clumps, cut-over areas, and adjacent uncut forest sites using point counts (Chapter 7). Bats will be surveyed using automatic remote detectors in residual clumps and in cut-over areas (Chapter 8)

Reporting

This report contains analyses for biota that were surveyed during 1998. We will prepare a final report incorporating all data by March 2000. The final report will include a summary chapter describing similarities and differences among biota groups that occurred as a consequence of retaining trees in many small clumps versus few large clumps at harvest.

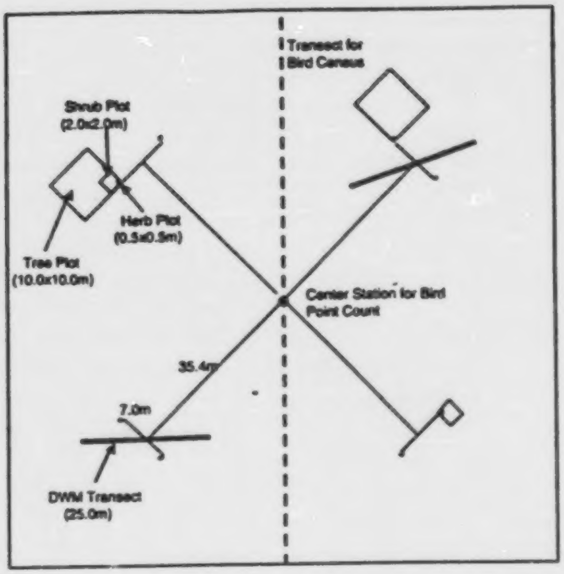
Transect for Bird Census
Shrub Plot (2.0x2.0m)
Herb Plot (0.5x0.5m)
Tree Plot (10.0x10.0m)
Center Station for Bird Point Count
35.4m
7.0m
DWM Transect (25.0m)

FIGURE 2. 3. An example of the nested-plot layout for vegetation and the bird transect within a 1-ha (2.47 acre) site pre-harvest during 1997. The sectors in which trees and shrubs were surveyed were randomly chosen within each 1-ha (2.47 acre) site.

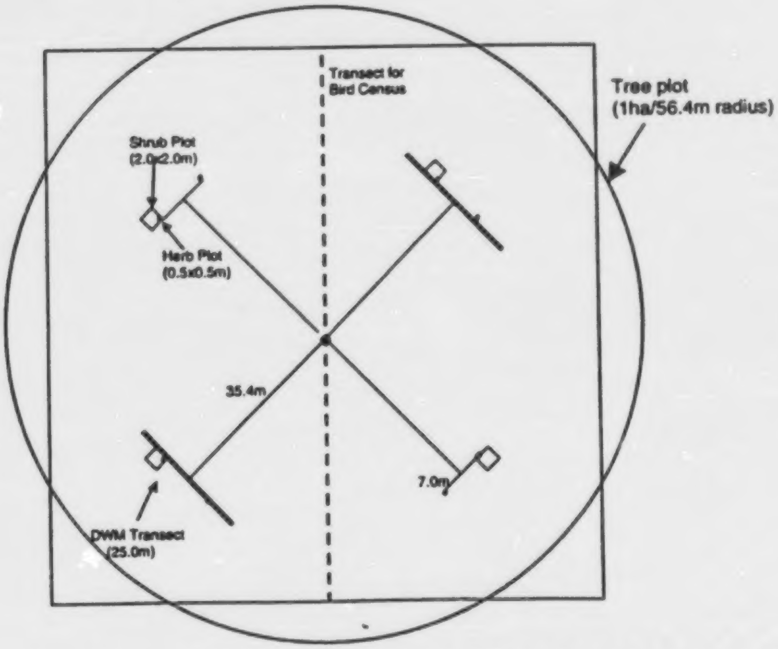
Transect for Bird Census
Tree plot (1ha/56.4m radius)
Shrub Plot (2.0x2.0m)
Herb Plot (0.5x0.5m)
35.4m
7.0m
DWM Transect (25.0m)

FIGURE 2.4. An example of the nested-plot layout for vegetation and the bird transect within a 1-ha (2.47 acre) site post-harvest during 1998.

TABLE 2.1. Schedule by which forest structure and biota were/will be surveyed during the project.

Element	1997 (Pre-Harvest)	1998 (Post-harvest)	1999 (Post-harvest)
Trees and Snags	extensive survey	extensive survey	cursory survey
Down woody Material	extensive survey	extensive survey	none
Understory Vegetation	cursory survey	none	extensive survey
Arthropods	none	extensive survey	none
Birds	extensive survey	cursory survey	extensive survey
Bats	extensive survey	extensive survey	extensive survey

CHAPTER 3. FALLDOWN OF TREES AND SNAGS IN RESIDUAL CLUMPS LEFT POST-HARVEST.

Susan Crites

Introduction

The natural disturbance model of ecosystem management assumes that biota are adapted to natural disturbances and are more likely to maintain viable populations if forest harvesting is as similar as possible to natural disturbances (Swanson and Franklin 1992). In Alberta's boreal mixedwood forests, fire is the dominant natural disturbance (Eberhart and Woodard 1987) and is currently being used as a template for timber harvesting by some companies (Anonymous 1997). Structure in the form of large trees and snags is often in abundance after fire, and its presence, mainly in the form of snags and downed woody material, is often seen long into succession (Lee et al. 1997). To allow timber harvesting to more closely approximate fire, forest managers are advocating that merchantable trees be left behind at harvest to serve as live trees, snags, and downed woody material as the stand ages. However, the appropriate amount and distribution of residual material on cutblocks has not been established. Clumps left behind at harvest could be compared to fire skips, and may serve as refugia and dispersal sites; therefore, larger clumps may function as better refugia (Saunders et al. 1991). Conversely, residual material evenly distributed throughout a cutblock could put more structure into post-harvest mature forests than is usually present. This structure could serve as habitat, cover, and foraging sites for other biota (Spies et al. 1991). Of the two patterns, big clumps versus scattered trees, it is not known which will be more beneficial to biota.

Trees in cutblocks that are newly exposed to the turbulent and accelerated winds are susceptible to blowdown because they have not developed the structural integrity to withstand such winds (Kimmins 1996). These trees grew within intact forests where they competed for light. It follows then that these trees tend to have thinner stems, more branches concentrated in the canopy, and poorly developed root systems compared to trees that have been exposed to wind during development (Kimmins 1996, Larsen 1963).

There are differences among tree species in susceptibility to blow down; however, many of these differences can be attributed to differences in structural integrity, decay properties, exposure to wind during development, and sheltering ability of surrounding vegetation. Deciduous species such as aspen and balsam poplar are more likely to snap off along the bole than to be root thrown. This is due to their concentration of branches in the canopy, the depth and complexity of their rooting system, and their susceptibility to fungal attack that weakens boles (Peterson and Peterson 1992, Putz et al. 1983). Coniferous species, because of their shallow rooting system and high surface area (consisting of needles and branches) exposed to wind that extends the entire length of the bole often are root thrown (Putz et al. 1983).

Characteristics of the soil and topography also influence the ability of trees to remain standing pre- and post-harvest. In shallow soils and/or soils saturated with water, root systems are poorly anchored and blowdown is more common (Kimmins 1996, Rodgers et al. 1995). In addition, frost heaving can uplift trees from wet soils. Season of harvest will influence root system stability and/or tree health in areas directly affected by equipment. Timber harvesting when the ground is not frozen can lead to soil compaction, and, if compaction is very high, it can kill root systems, leading to blowdown and/or tree death (Kimmins 1996).

In this report I present a general description of the experimental clumps left after harvest. In addition, I determine tree and snag fall down rates over one growing season and whether falldown rates vary among treatments.

Methods

For a description of study areas and methods, see Chapter 2. Cutblocks were harvested during the winter of 1997/98. Trees were surveyed in a randomly selected subset of clumps. Within each stand group, in both Whitecourt and Grande Prairie, we selected one large clump, four medium clumps, and 16 small clumps, to be surveyed. Thus four large, 16 medium, and 64 small clumps were surveyed in each study area. In addition, two 1-ha (2.47 acre) sites were sampled in cutover areas adjacent to each of the three clump sizes, in each of the four stand groups, totaling 24 cutover sites. Eight one ha (2.47 acre) sampling sites were also placed in the adjacent unharvested forest in each study area. Below these sampling units (small, medium, and large clumps and cutover and forest areas) will be referred to as habitat types.

Time and weather permitting, habitat types were sampled immediately post-harvest or as soon as possible in the spring. In addition, all habitat types were re-visited during the first growing-season post-harvest to determine the rate of falldown. In cases where the clumps could not be accessed immediately post-harvest, they were sampled during the spring and summer, 1998. In these clumps downed trees were evaluated to determine whether they fell pre- or post-harvest by evaluating whether the downed tree had been run over by equipment or if there was any vegetation growth on the tree that had obviously come from the ground. If a tree fell post-harvest, both its pre-and post-falldown characteristics were recorded.

Each habitat type was sectioned into four quadrants (I = 1-90°, II = 91-180°, III = 181-270°, and IV = 271-360°) based on ordinal directions. All trees and snags were surveyed beginning in Quadrant I, and continued in a clockwise direction to quadrant IV. All trees and snags ≥10.0 cm (4 in.) in diameter at breast height [dbh, 1.4 m (4.6 ft.)] were tagged with a numbered aluminum tree tag on the north side of the tree at approximately 1.7 m (5.6 ft.) from the ground. Tags were nailed into the trees, leaving the tag lose for continued tree growth. Variables measured for each tagged tree/snag included species, dbh, height class [1 = 1.4-5 m(4.6-16.4 ft.), 2 = 5.1-10 m (16.5-32.8 ft.), 3 = 10.1-20 m 32.9-65.6 ft.), 4 = >20 m(>65.6 ft.)], condition (alive or dead), top condition (intact, broken canopy, broken bole), and whether conks (fungal fruiting bodies) were present or absent. If the tree was dead, the decay stage was also recorded (six stages from Lee et al. 1997; stage one being a recently killed tree, stage six being a broken off, well-decayed bole). Much of these data were collected so that we will have an initial record to which we can compare future changes. Therefore some of these data (e.g., top condition, fruiting bodies, decay stages, height classes) will not be presented in this report.

During the revisit to each habitat type, tree tags were recorded along with species, condition, position (downed or standing), height class, top condition, and direction the bole was facing if the tree was down. It was possible to have a downed live tree, where the leaves or needles were green. These trees were recorded as recently fallen.

Data analysis

Most data were analysed using multinomial loglinear analysis with tree fate (i.e., standing or fallen) as the dependant variable. G^2 statistics were used, as they are preferred over Chi square tests (Zar 1984). Testing was completed for overall habitat types (small, medium, large) and whether the trees within each habitat type were inside or outside a clump (i.e., in the clump (In), or in the cutover area (Out)) as the independent variables and falldown rates as the dependent variable (SPSS 7.5 1997). If there were significant interactions between habitat types and location In/Out of clumps (Habitat Type*In/Out), then habitat types and In/Out were examined separately, otherwise, no further comparisons were made. Total stand falldown was examined by calculating a weighted percentage of falldown for each of the three habitat types (falldown rate for trees in clumps times the proportion of trees in clumps plus falldown rate for trees in cutover times proportion of trees in cutover). Dbh of fallen trees was analysed using logistic regression (SPSS 7.5 1997) to determine whether falldown rates were related to tree diameter. All results were considered significant if $p \leq 0.05$

Results

Clump Characteristics

Regardless of habitat type, the number of trees remaining in a clump was highly variable in both study areas. Tree densities were higher per habitat type in Whitecourt than in Grande Prairie (Tables 3.1 and 3.2), probably because the dominant trees species differed between study areas. Dbh and tree spacing also differed between the two areas. Tree species composition was similar to the composition pre-harvest (see below and Progress Report 1998); however, there was difficulty for field staff distinguishing black spruce from white spruce, so these species were grouped in analyses.

Falldown Rates by Habitat Type

Overall falldown rates were 2.8% and 8.9% in Grande Prairie and Whitecourt, respectively. There were significant interactions between habitat type (i.e., small, medium, and large) and whether trees were inside or outside the clumps (i.e., in the cutover or inside the clump), therefore falldown rate was examined separately for each factor (Table 3.3; note there was no falldown in the unharvested forest sites and these were not included in analyses). When habitat types (i.e., falldown rates of trees within the three clump sizes) were examined separately from cutover areas, there were significant differences among clump sizes for both study areas (Table 3.3; Figures 3.1 and 3.2). In addition, when cutovers were examined separately from treatments, falldown rates were significantly different among trees outside of small, medium, and large clump types in the Grande Prairie study area, but these differences were not significant for the Whitecourt study area (Table 3.3).

When falldown rates inside and outside the clumps were compared within a single habitat type, in both study areas, trees remaining in cutovers adjacent to large clumps had significantly higher falldown rates than trees within the clump. The two study areas responded differently for small and medium habitat types. In the Grande Prairie study area, the falldown rates inside small and medium clumps were not significantly different from the falldown rates in the respective small and medium cutovers (Table 3.3, Figure 3.1). In the Whitecourt study area, falldown rates were significantly higher within small and medium clumps, than in the respective cutover areas (Table 3.3, Figure 3.2). The weighted total falldown for each habitat type was similar to the falldown within clumps due to the low original tree densities in the cutovers compared to falldown rates in clumps (Figures 3.1 and 3.2).

Falldown Rates for each Species

Falldown rates for individual tree species were calculated by dividing the number of trees fallen by the original number of trees present, then converting the result to a percentage. In the Grande Prairie study area, the dominant tree species was aspen followed by balsam poplar, then spruce (as mentioned in Methods section, no distinction was made between white and black spruce, Table 3.4). Spruce had the highest relative falldown rate (17.9%) in the Grande Prairie study area. The Whitecourt study area had lodgepole pine, followed by spruce, then paper birch, and aspen for dominant tree species (Table 3.4). Balsam poplar had the highest relative falldown rate in Whitecourt, but there were only 11 trees surveyed, therefore, spruce was a better representative of highest falldown rate (15.6%) (Table 3.4).

When species relative falldown rates were examined for each habitat type, spruce had the highest falldown rates in both study areas. In Grande Prairie, spruce had a falldown rate of 21.4% in small clumps, followed by balsam poplar at 4.6% (Figure 3.3). Spruce falldown rates were 10.4%, 2.9%, and 26.1% in medium, large, and cutover habitat types, respectively (Figure 3.3). In Whitecourt, spruce had falldown rates as high as 24.7% in medium clumps, but balsam poplar and lodgepole pine had the highest falldown rates in the cutover areas (27.3% and 22.2%, respectively; Figure 3.4). Falldown rates for the latter two species were based on three and two trees falling down out of a pre-treatment density of 11 and nine, respectively; therefore, paper birch is a better representative of falldown rates in cutovers having 6% (22) of the 368 trees fall (Figure 3.4).

Falldown Rates Within each of the Four Quadrants

The predominant wind direction in both study areas is from the west, and occasionally from the northwest in Whitecourt (Canadian Climate Normals 1993). I determined whether trees in

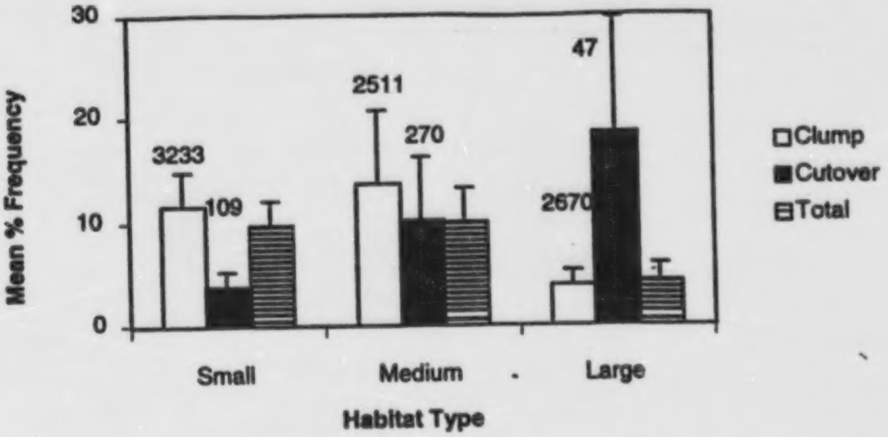

FIGURE 3.2. Average (±SE) percentage falldown of trees in small, medium, and large clump types, the respective adjacent cutover areas, and the weighted total for each habitat type for the Whitecourt study area. Numbers above bars indicate surveyed number of trees.

TABLE 3.4. Total number of trees, density of fallen trees, and falldown rates (%) for each species during first growing season in Grande Prairie and Whitecourt study areas.

	Grande Prairie			Whitecourt		
	Original No. Trees	No. Trees Down	Falldown Rate (%)	Original No. Trees	No. Trees Down	Falldown Rate (%)
Aspen	5944	134	2.3	528	21	4.0
Balsam Poplar	791	17	2.2	11	3	27.3
Paper Birch	64	1	1.6	798	53	6.6
Spruce*	257	46	17.9	2512	393	15.6
Lodgepole Pine	4	0	0	4832	299	6.2
Balsam Fir	1	0	0	158	15	9.5

*White and black spruce combined.

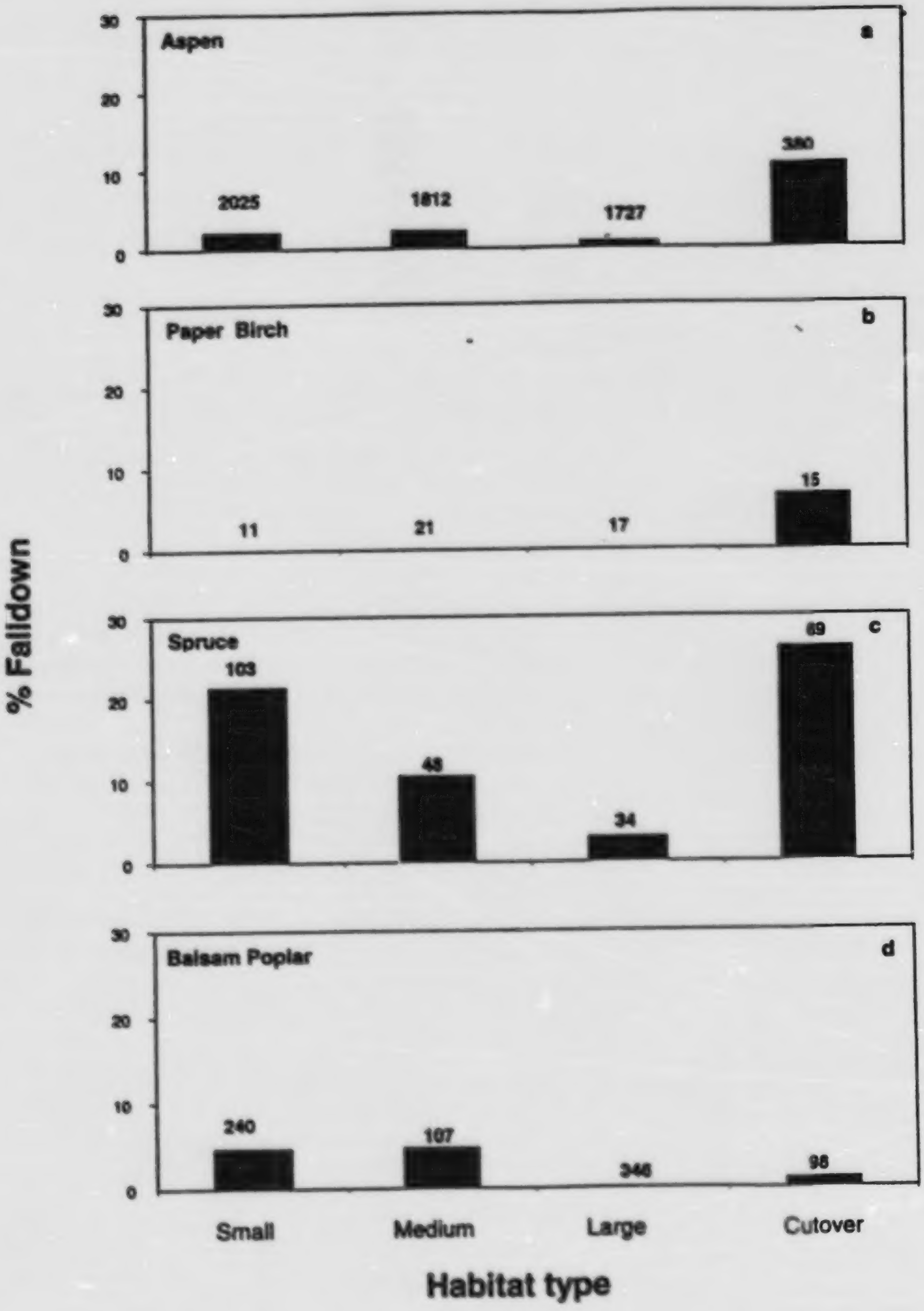

FIGURE 3.3. Percentage of tree falldown categorized into habitat types of small, medium, large, and cutover for each a) aspen, b) paper birch, c) spruce (black and white), d) balsam poplar for the Grande Prairie study area. Numbers above bars indicate number of trees surveyed.

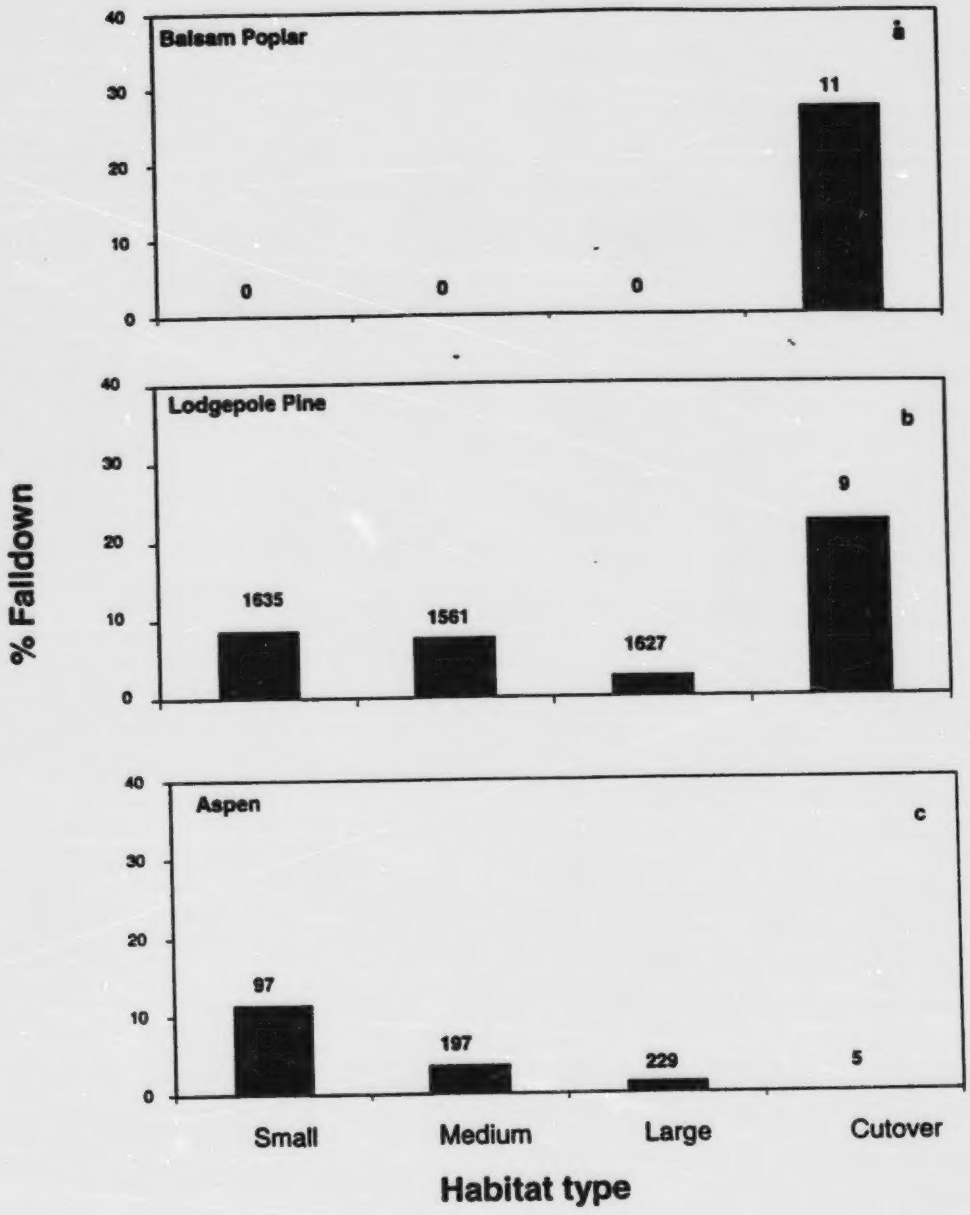

FIGURE 3.4. Percentage of tree falldown categorized into clump types of small, medium, large, and cutover for each a) balsam poplar, b) lodgepole pine, c) aspen, d) spruce (black and white), e) balsam fir, f) paper birch for the Whitecourt study area. Numbers above bars indicate number of trees surveyed.

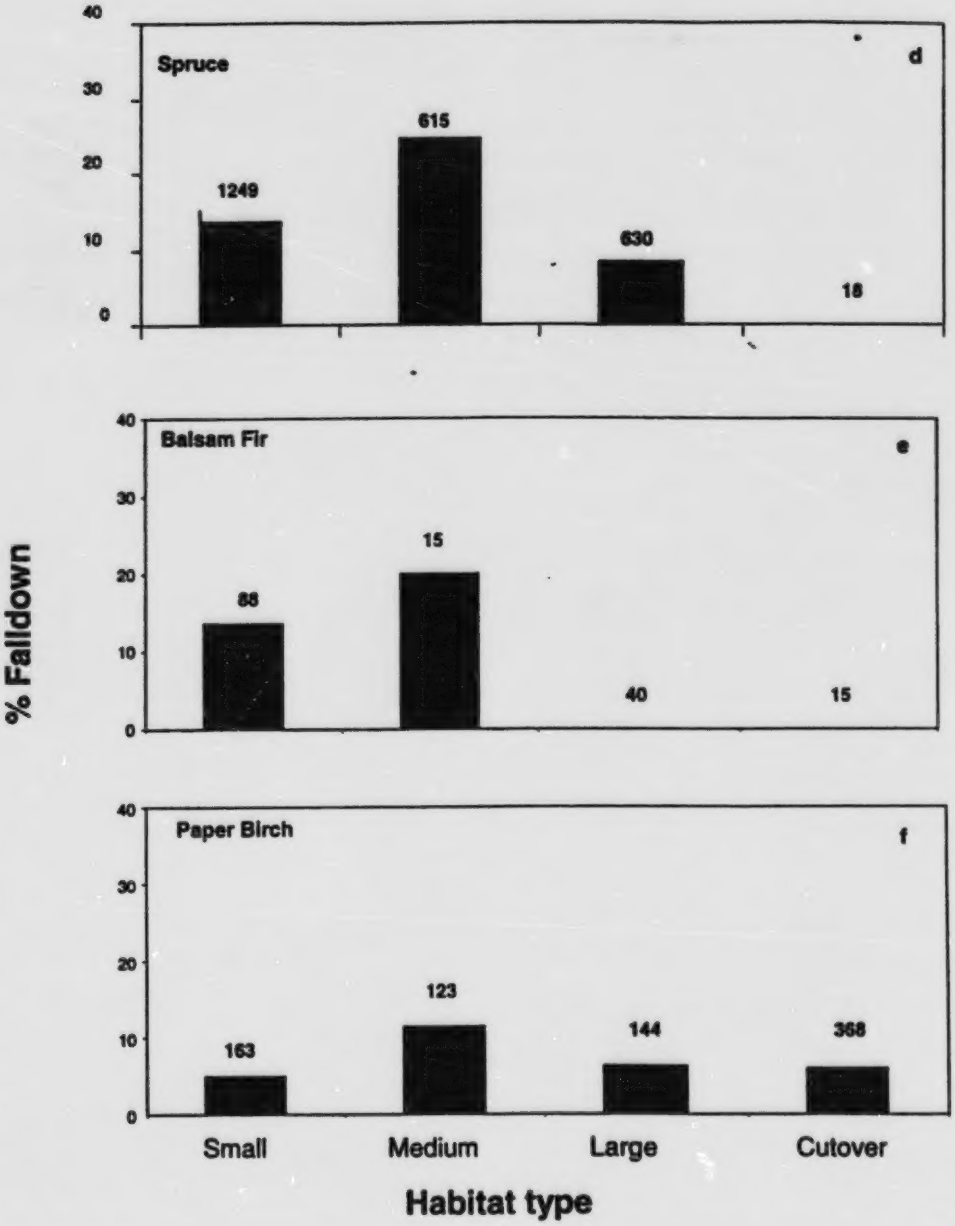

FIGURE 3.4. cont. Percentage of tree falldown categorized into clump types of small, medium, large, and cutover for each a) balsam poplar, b) lodgepole pine, c) aspen, d) spruce (black and white), e) balsam fir, f) paper birch for the Whitecourt study area. Numbers above bars indicate number of trees surveyed.

the northwest and west quadrants had higher falldown rates than trees in other quadrants, for all clump sizes combined and within each of the habitat types separately. There was no relationship in Grande Prairie between falldown rate and quadrant ($G^2 = 2.6$, df = 3, $p = 0.465$; Figure 3.5); however in Whitecourt, falldown rate was higher in quadrants 181-270° and 271-360° ($G^2 = 18.2$, df = 3, $p < 0.0001$; Figure 3.6). This pattern was most obvious in large clumps, and was not present in small and medium clumps. The size of large clumps is likely responsible for the differences among clump size since small and medium clumps are too small to have a quadrant effect. Within cutovers there is no relationship in either study area between quadrant and falldown rate, likely because there is no consistent windward side in openings (Figures 3.5 and 3.6).

Direction of Falldown

Trees tended to fall in between 1-180° in the Grande Prairie study area ($G^2 = 86.2$, df = 3, $p < 0.0001$; Figure 3.7), and between 91-180° in the Whitecourt study area ($G^2 = 666.4$, df = 3, $p < 0.0001$; Figure 3.8).

Relationship Between Falldown and Dbh

In both study areas, there was a negative relationship between tree size and probability of falling down ($G^2 = 17.6$, df = 1, $p = 0.0000$, $G^2 = 106.7$, df = 1, $p = 0.0000$ for Grande Prairie and Whitecourt, respectively). In Grande Prairie, average dbh was 22.7 cm (8.9 in.) and 21.3 cm (8.4 in.) for standing and fallen trees, respectively. In Whitecourt, the average dbh of standing trees and fallen trees was 16.5 cm (6.5 in.) and 18.7 cm (7.4 in.), respectively. Trees in cutovers were larger on average, than trees in clumps for both study areas (Grande Prairie: mean dbh of 22.5 cm (8.9 in.) and 25.9 cm (10.2 in.) for clumps and cutovers, respectively; Whitecourt: mean dbh of 18.6 cm and 20.0 cm for clumps and cutovers, respectively); therefore the high falldown rates in cutovers are not due to the presence of small trees.

Discussion

With mean overall falldown rates as low as 2.8% and 8.9% in Grande Prairie and Whitecourt, respectively, the vast majority of trees remained standing, regardless of habitat type. The lower overall falldown rates in Grande Prairie compared to Whitecourt may be due to Grande Prairie having drier sites, less site to site variability, an interspersion of cutover and leave areas, and most probably, aspen as the dominant tree species. Trees rooted in poorly drained soils are more likely to falldown when exposed to wind than trees rooted in well drained soils (Kimmins 1996, Lohmander and Helles 1987). We know from the tree species present in each study area, and from personal observation, that the Grande Prairie study area had soils that were drier overall than soils in the Whitecourt study area. In addition, the distribution of cutover and leave areas in the Grande Prairie study area allowed for more wind buffering than occurred in Whitecourt where cutovers were located adjacent to one another (Figures 2.1 and 2.2). Such large openings, as those present in the Whitecourt study area, may facilitate accelerated and turbulent winds (Kimmins 1996), leading to higher falldown rates than in the Grande Prairie study area.

Average annual windspeeds were actually higher in Grande Prairie than in Whitecourt (Canadian Climate Normals 1993), which does not serve to explain the results. Aspen is generally considered a relatively windfirm species (Jones and DeByle 1985) and is better than white spruce at withstanding wind due to its limited bole branching and its deep rooting system. Spruce, on the other hand, is characterised by bole branching and a shallow rooting system making it susceptible to windthrow (Kimmins 1996, Putz et al. 1983). Bole branching, along with many fine needles, increases the surface area exposed to wind, acting much like a sail at intercepting wind and facilitating blowdown. In addition, the shallow roots of white spruce make it more susceptible to ground-level disturbances such as compaction due to logging equipment, or frost heaving (Kimmins 1996).

The blowdown that occurred in aspen can be explained by decay and canopy configuration. Fungi often decay aspen boles and roots, and the canopy branching is often dense, widely spreading, and heavy, making aspen susceptible to snapping along the bole in response to wind

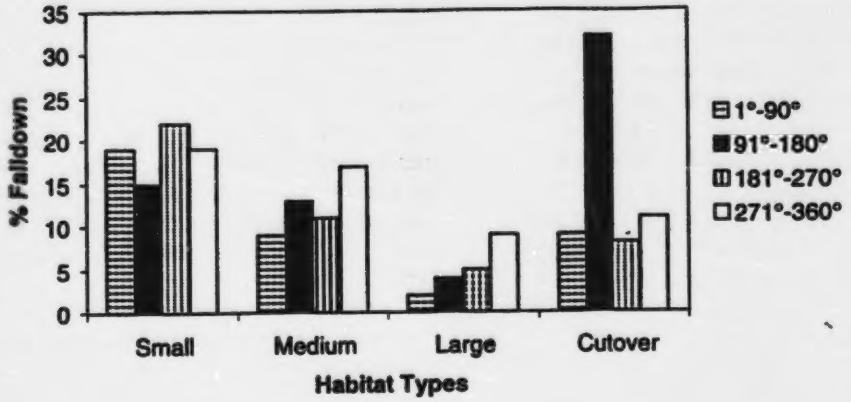

FIGURE 3.5. Percent of trees that fell in four different ordinal sampling quadrants for small, medium, large, and cutover habitat types in the Grande Prairie study area.

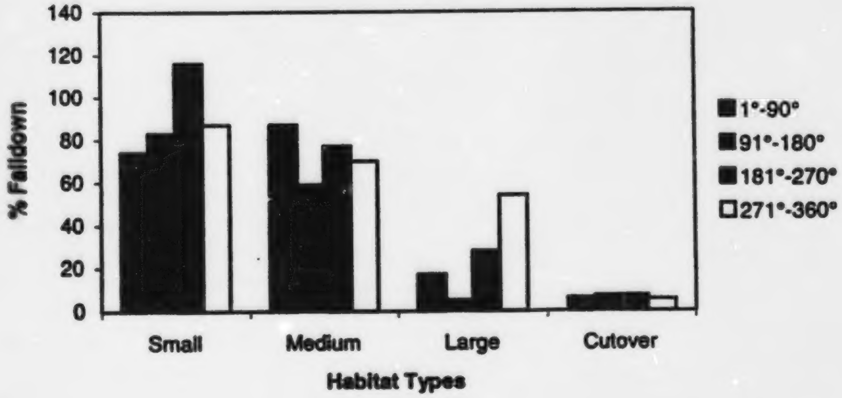

FIGURE 3.6. Percent of trees that fell in four different ordinal sampling quadrants for small, medium, large, and cutover habitat types in the Whitecourt study area.

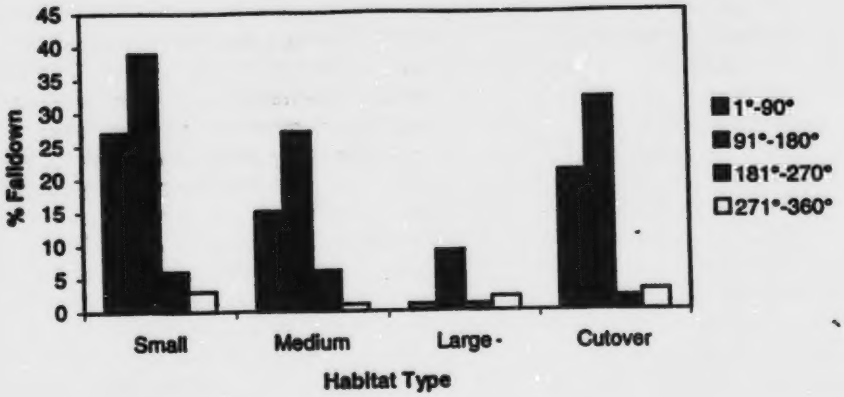

FIGURE 3.7. Percent of trees that fell in each of four ordinal direction categories for small, medium, large, and cutover habitat types for the Grande Prairie study area.

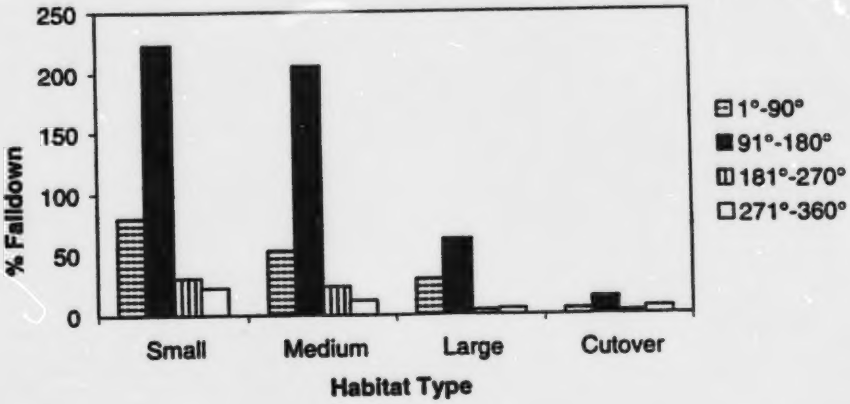

FIGURE 3.8. Percent of trees that fell in each of four ordinal directions for small, medium, large, and cutover habitat types for the Whitecourt study area.

stress and decay (Jones and DeByle 1985, Putz et al. 1983). The results of this study indicate that this susceptibility is increased when aspen is left isolated in cutover areas, than left in clumps.

Lodgepole pine was well represented in the Whitecourt study area, but experienced relatively low falldown rates. Little literature is available on blowdown in pine, but personal observation indicates that pine has a different structural conformation than spruce; it tends to have lighter and fewer branches, and more loosely packed needles compared to white spruce. It also undergoes self-pruning therefore does not often branch as far down the bole as white spruce. These factors combined allow for wind flow through the tree and its branches, rather than acting as an impediment (Lohmander and Helles 1987).

In both study areas falldown rates were lowest in large clumps compared to small and medium clumps. Large clumps provide shelter to more trees from prevailing winds, have reduced soil disturbance per unit area, and have an intact interior understory that is not directly influenced by introduced elements such as wind and radiation. Because wind velocity decreases as it moves inward through the forest from the forested edge (Kimmins 1996, Lohmander and Helles 1987), small and medium clumps were more influenced by wind throughout the clump.

Isolated trees left in cutover areas were more likely to remain standing if the surrounding cutblock was composed of small and medium clumps, rather than large clumps. If wind velocity becomes accelerated in large openings (Kimmins 1996), it follows that a cutblock containing few large clumps will experience less wind buffering than a cutblock with many small or medium clumps dispersed throughout.

Another reason for the high falldown of isolated trees in cutovers is that trees that have been historically surrounded by an intact forest will be more susceptible to blowdown than trees that have historically been isolated from surrounding forest. This occurs because during competition for light, forest trees tend to grow tall and slender with shallow roots (Kimmins 1996). Trees growing out in the open, on the other hand, do not have to grow tall for light, so they put more resources into increasing bole diameter and establishing a deeper, more developed rooting system (Kimmins 1996). When exposed to wind, unsheltered trees will receive bending stimuli which promotes the development of strong supporting roots (Gratkowski 1956). It follows then, that if the surrounding shelter is suddenly removed from a tree, as is done at harvest, it will be weaker structurally, and will be more likely to blow down than a tree that has been historically isolated. The results of this study support this as fallen trees had significantly smaller diameters than standing trees. Navratil et al. (1994) demonstrated that structural adaptations to wind occur within five years of exposure, resulting in lower falldown rates with time since harvesting.

Wind was moderate (average annual wind speeds of 12 km/h (7.5 mph) and 8 km/h (5 mph) for Grande Prairie and Whitecourt, respectively) in both study areas (Canadian Climate Normals 1993). As expected, in both study areas, within small and medium clumps similar numbers of trees fell on the windward and leeward sides. These clumps were likely too small and contained too few trees to significantly buffer winds coming into the clump. In addition, wind can become turbulent on the leeward side of a clump, thus potentially pulling trees down on the lee side (Kimmins 1996). However, for large clumps in the Whitecourt study area, higher falldown occurred on the windward side (the direction of the prevailing wind is primarily from the west in Grande Prairie and from the west and northwest in Whitecourt, Canadian Climate Normals 1993). This pattern was not statistically significant in large clumps in the Grande Prairie study area, likely due to the small sample size of fallen trees (n=13). Future revisits will provide a better sample size to examine this relationship further.

Summary

After one growing season, overall falldown rates were low for both Grande Prairie and Whitecourt study areas. Falldown rates were lowest in large clumps and highest in cutover areas adjacent to large clumps. Falldown rates varied by tree species, with aspen being the most

Rodgers, M.A., Casey, A., McMenamin, C., and Hendrick, E. 1995. An experimental investigation of the effects of the dynamic loading on coniferous trees planted on wet mineral soils. *In* Wind and Trees. *Edited by* M.P. Coutts and J. Grace pp. 201-219.

Saunders, D.A., Hobbs, R.J., and Margules, C.R. 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology 5: 18-32.

Spies, T.A., Tappeiner, J., Pojar, J., and Coates, D. 1991. Trends in ecosystem management at the stand level. *In* Transactions of the fifty-sixth North American wildlife natural resources conference. *Edited by* R.E. McCabe, Wildlife Management Institute, Washington DC. pp. 628-639.

SPSS Inc. 1997. SPSS 7.5.1 for Windows.

Swanson, F.J. and Franklin, J.F. 1992. New forestry principles from ecosystem analysis of Pacific Northwest forests. Ecological Applications 2: 262-274.

Zar, J.H. 1984. Biostatistical Analysis, 2nd ed. Prentice-Hall, Inc., New Jersey.

windfirm and spruce the most susceptible to wind. Falldown rate also was negatively related to tree dbh, and higher on the windward side of clumps (significant for Whitecourt only). It would be informative to evaluate whether falldown rates were related to topographic relief, soil hydrology, and soil characterisation of the tree clumps.

Management Implications

Large clumps provided the most suitable configuration for maintaining standing residual trees in a cutblock. The downside of leaving only large clumps is that any residual material remaining outside of the clump has an increased likelihood of falldown. In addition, there may be trade-offs associated with other biota that make few large clumps in a cutblock too limiting for dispersal to adjacent large clumps and intact forest. Perhaps a configuration that would leave large clumps in the most exposed locations with medium or small clumps dispersed among them and between the clumps and the block boundary would be a valuable solution. Due to the higher falldown rates, leaving isolated residual material in cutover areas probably is of little value for all except aspen trees. If managers have a choice of which trees or stands to leave as residual material, larger trees should be chosen since trees with larger diameters were more likely to remain standing than are trees with smaller diameters. This study also demonstrated that spruce has the highest falldown rate of any other species, and perhaps this species should not be left as residual material unless it is of sufficient diameter or part of a large clump configuration. The trade-off for spruce contributing a valuable seed source must be evaluated.

Literature Cited

Anonymous, 1997. Alberta Pacific Forest Industries, Inc. Detailed Forest Management Plan. 143 pp.

Eberhart, K.E. and Woodard, P.M. 1987. istribution of residual vegetation associated with large fires in Alberta. Canadian Journal of Forest Research 17: 1207-1212.

Gratkowski, H.J. 1956. Windthrow around staggered settings in old-growth Douglas-fir. Forest Science 2: 60-74.

Jones, J.R. and Debyle, N.V. 1985. Morphology. *In* Aspen: ecology and management in the western United States. *Edited by*: N.V. DeByle and R.P. Winokur. U.S. Dep. Agric., For. Serv., Rocky Mt. For. Range Exp. Stn., Fort Collins, CO. Gen. Tech. Rep. RM-119.

Kimmins, J.P. 1996. Forest Ecology: A Foundation for Sustainable Management. 2nd ed. Prentice Hall, Upper Saddle River, NJ. 596 pp.

Larsen, P.R. 1963. Stem form development of forest trees. For. Sci. Monogr. 5: 1-42.

Lee, P.C., Crites, S., Nietfeld, M., Van Nguyen, H. and Stelfox, J.B. 1997. Characteristics and origins of deadwood material in aspen-dominated boreal forests. Ecological Applications 7: 691-701.

Lohmander, P. and Helles, F. 1987. Windthrow probability as a function of stand characteristics and shelter. Scand. J. For. Res. 2: 227-238.

Navratil, S., Brace, L.G., Sauder, E.A., and Lux, S. 1994. Silvicultural and harvesting options to favor immature white spruce and aspen regeneration in boreal mixedwoods. Natural Resources Canada, Canadian Forest Service, Northwest Region, Northern Forestry Centre, Edmonton, AB. Info. Rept. NOR-X-327.

Peterson, E.B., and Peterson, N.M. 1992. Ecology, management, and use of aspen and balsam poplar in the prairie provinces, Canada. For Can, Northwest Reg, North For Cent, Edmonton, Alberta. Spec Rep 1.

Putz, F.E., Coley, P.D., Lu, K., Montalvo, A., and Aiello, A. 1983. Uprooting and snapping of trees: structural determinants and ecological consequences. Can. J. For. Res. 13: 1011-1020.

CHAPTER 4. CHARACTERISTICS OF DOWNED WOODY MATERIAL IN RELATION TO THE DISPERSION OF HARVEST RESIDUAL

Troy C. Sorensen

Introduction

Bryophytes, lichens, fungi, invertebrates, and herbaceous plants compose the majority of biodiversity in an ecosystem. Many of these taxa are influenced by, or dependent on, the availability of down woody material (DWM) substrate (McCullough 1948; Thompson 1980; Söderström 1988ab; Amaranthus et al. 1994; Crites and Dale 1995, 1998). DWM is also crucial as nest sites (Thompson 1980), food, and protection for vertebrates and invertebrates (Harmon et al. 1986, Lee et al. 1995). In addition to its direct influence on biodiversity, DWM functions as an important reservoir for ecosystem nutrients and contributes to ecosystem resilience (Lambert et al. 1980; MacMillan 1981; Abbott and Crossley 1982; Harmon et al. 1986).

Large DWM in a wide range of decay stages is often stressed as important in maintaining DWM-dependent taxa (Söderström 1988a; Crites and Dale 1998); however, harvesting reduces the size distributions (Abbott and Crossley 1982; Lee et al. 1995) and decay class distributions of DWM (Söderström 1988b). One strategy to minimize these effects is to leave residual live trees, snags, and DWM in harvested areas (Lee et al. 1995) in an effort to promote natural size and decay class distributions of DWM, and the establishment of DWM-dependent taxa (Söderström 1988b). If forest managers are willing to leave this residual material, the question remains of how to distribute it within harvested areas. An even spatial distribution of DWM will facilitate the dispersal of DWM-dependent species, but will minimize the amount of interior-forest environment which promotes the even decay class distribution of DWM.

This study has four objectives:

1. To describe patterns of decomposition in aspen and lodgepole pine DWM in the Grande Prairie and Whitecourt areas, respectively.
2. To provide pre-harvest information on DWM in the two study areas and determine the influence of forest harvest on DWM.
3. To determine the influence of residual material dispersion on the characteristics of DWM in harvested stands of the two study areas.
4. To determine if the residual clumps of DWM in harvested stands are representative of the pre-harvest DWM characteristics.

Methods

A description of the study areas and experimental harvesting is presented in *Study Area and Methods* (Chapter 2). Ten 1-ha (2.47 acre) *sites* were established in the center [*ca.* 100 m from the edge (110 yd.)] of each stand and all surveys for biota were conducted within those sites (Fig. 2.1 and 2.2). Down woody material was surveyed in the summer of 1997 (pre-harvest) and again in 1998 (post-harvest). Down woody material was surveyed along two 25 m (27 yd.) transects within the center of two randomly selected quadrants of each site (Fig. 2.3). The direction of each transect was random. All DWM intersecting the transect with 5-10 cm (2-4 in.) diameters were tallied. The following was recorded for pieces > 10 cm (> 4 in.) at transect intersection: diameter, species, decay stage (Table 4.1), wood state (Table 4.2), percent of bark cover within 1m (3 ft.) of intersect, percent vegetation cover within 1m (3 ft.) of intersect, condition (broken or intact bole), whether it was a stump, and position (raised or on the ground). Any stump that was less than 1.5 m (5 ft.) tall, or any fallen snag at an angle less than 45 degrees, was also counted as DWM. From hereon, small and large DWM will refer to DWM with diameters between 5-10 cm (2-4 in.) and >10 cm (4 in.) respectively.

TABLE 4.1. Descriptions of decay classes (modified from Lee et al. 1997) assigned to large [>10 cm (>4 in.) diameter] DWM.

Decay Classes	DWM Characteristics
1	Log whole and undecayed; bark, branches, and twigs present and intact; log elevated on support points; wood hard.
2	Log sound, wood hard; twigs mostly lacking; <50% of the bark missing
3	Wood soft in places; some branches remaining; >50% bark missing
4	Little to no bark remaining; no branches; wood fungi present; wood soft with small crevices and small pieces lost.
5	Large wood fragments lost; outline of trunk slightly deformed; vascular and non-vascular plants beginning to colonize.
6	Wood mostly well-decayed; log colonized by vascular and non-vascular plants.
7	Humidification nearly 100%; hard to define as log, outline indeterminable; no evidence of hard wood.

TABLE 4.2. Descriptions of wood states assigned to large [>10 cm (> 4 in.) diameter] DWM.

Wood State	Wood Description
1	Hard and intact
2	Top 1cm flaking or spongy
3	Top 2-3 cm (0.8-1.2 in.) fragmenting, semi-hard core
4	Entire log is highly fragmented and easily falls apart
5	Retains outline of log but is nearly humus.

In the summer of 1998, additional transects were sampled at random directions through the middle of residual tree clumps for a total of 80 m (88 yd.) per stand; i.e. 80 m (88 yd.) through 1 large clump, 40 m (44 yd.) through 2 medium sized clumps, and 20 m (22 yd.) through 4 small clumps. These samples were used to characterize the residual clumps and to determine any differences between the clumps and the characteristics of pre-harvest DWM. The volume of DWM per unit area (v) was calculated from the following formula (Van Wagner 1968):

$$v = \frac{\pi^2 \sum_{i=1} d_i^2}{8l}$$

where l is the transect length [25 m (27 yd.)], d is the diameter (cm) of the large pieces (>10cm) or 7.5 cm (3 in.) for small pieces [5-10 cm (2-4 in.)], and i is the number of pieces intersecting the transect. The resulting units are cubic meters per hectare. Volumes were calculated for both small and large pieces and then summed for a total value. Volumes were also calculated for each decay class (1-7) of the large pieces.

To calculate biomass estimates from volume estimates it was necessary to determine the wood density of DWM within each study area. Methods to sample DWM wood density were taken from Foster and Lang (1982). Cross-sections of aspen and lodgepole pine DWM were collected from the Grande Prairie and Whitecourt study areas respectively. Five samples were collected from each of decay classes 1-6 (decay class 7 was too decomposed to consistently distinguish woody material from humus and soil). Cross-sections were 5cm thick and taken from logs with 15-20 cm (6-8 in.) diameters. Each cross-section was measured for volume, weighed, dried at 70°C (158 F) in a forced-air dryer, and re-weighed. Wood density was calculated as the dried weight divided by the pre-dried volume. Moisture content was calculated from the difference between the dried and pre-dried weights. Biomass estimates of DWM within stands was calculated as the product of the volume estimate and the wood density estimate for each decay class. Wood density estimates for decay class 6 were also used for decay class 7; this will result in slight overestimates of biomass for decay class 7 DWM.

Data Analysis

General Linear Models (GLM, SPSS 1997) were used to examine differences in the volumes of DWM between pre-harvest and post-harvest stands, between stand treatments, and between the pre-harvest stand and the characteristics of residual clumps. Volumes of small DWM, large DWM, and total DWM were analyzed separately. Tukey's Honestly Significant Difference tests (p= 0.05) were used for all post hoc tests (Zar 1984; SPSS 1997).

Decay classes did not always accurately describe DWM, therefore other observable characteristics were analyzed to explain the underlying pattern of decomposition. Decay classes 1-7 were used as samples in a detrended correspondence analysis (DCA) with percent bark cover, percent vegetation cover, and wood state as variables. The DCA was conducted using aspen in the Grande Prairie study area and lodgepole pine in the Whitecourt study area.

Results

Grande Prairie Area

Post-harvest volumes of small and large DWM were significantly higher than pre-harvest volumes (small DWM, $F_{1,456} = 132.2$, $p < 0.001$; large DWM, $F_{1,456} = 40.4$, $p < 0.001$; total DWM, $F_{1,456} = 73.6$, $p < 0.001$; Table 4.3). The volume of large DWM and total DWM was highest in stands with small clumps (large DWM, $F_{2,8} = 9.0$, $p = 0.01$; total DWM, $F_{2,8} = 9.8$, $p = 0.01$); however, the volume of small DWM was similar among all stand treatments ($F_{2,8} = 0.3$, $p = 0.74$, Table 4.4). The volume of DWM within clumps after harvest was similar to the volume prior to harvest (small DWM, $F_{1,255} = 1.0$, $p = 0.33$, large DWM, $F_{1,255} = 0.02$, $p = 0.89$; total DWM, $F_{1,255} = 0.001$, $p = 0.97$, Table 4.3).

There was 12.4 Mg/ha (± 0.7 SE) [5.5 ton/acre (±0.3 SE)] of large DWM prior to the harvest of stands in the Grande Prairie area. This biomass was almost 3 times higher after harvest [32.5 ± 1.7 Mg/ha (14.5 ± 0.8 ton/acre), $F_{1,456} = 131.2$,

TABLE 4.3. Average volume (± SE) of DWM within stands during both years in both study areas. The number of transects sampled (n) is provided. Columns with similar superscripts are not significantly different ($\alpha = 0.05$).

			Volume (m^3/ha)		
	Year	n	Small DWM	Large DWM	Total DWM
Grande Prairie	Pre-Harvest	240	13.8 (0.6)^a	63.6 (3.4)^a	77.4 (3.4)^a
	Within Clumps Post-Harvest	28	15.8 (2.0)^a	61.2 (7.8)^a	77.0 (8.6)^a
	Post-Harvest	240	27.9 (1.2)^b	99.7 (5.1)^b	127.5 (5.5)^b
Whitecourt	Pre-Harvest	240	15.4 (0.7)^a	155.2 (7.9)^a	170.6 (7.9)^a
	Within Clumps Post-Harvest	28	19.6 (2.3)^a	138.2 (20.4)^a	157.9 (20.6)^a
	Post-Harvest	240	30.1 (1.2)^b	134.8 (6.2)^b	164.9 (6.1)^a

TABLE 4.4. Average volume (± SE) of DWM within stand treatments in both study areas. The number of transects sampled (n) is provided. Pre-harvest volumes were used as covariates so post-hoc test could not be done. Significant tests are denoted with ** ($p \leq 0.05$) or * ($p \leq 0.10$).

			Volume (m^3/ha)		
	Treatment	n	Small DWM	Large DWM	Total DWM
Grande Prairie	Large Clumps	80	24.0 (2.3)	80.4 (7.9)	104.4 (8.8)
	Medium Clumps	80	32.2 (2.4)	97.9 (7.3)	130.1 (7.9)
	Small Clumps	80	27.4 (1.5)	120.6 (10.6)	148.1 (10.9)
			n.s.	**	**
Whitecourt	Large Clumps	80	31.4 (2.2)	107.1 (9.4)	138.6 (9.2)
	Medium Clumps	80	29.7 (1.9)	133.4 (9.4)	163.2 (9.2)
	Small Clumps	80	29.0 (2.2)	163.8 (12.5)	192.8 (12.4)
			n.s.	*	**

$p < 0.001$]. The biomass of large DWM was higher in stands with small clumps [39.5 ± 3.4 Mg/ha (17.6 ± 1.5 ton/acre)] than in stands with medium [34.2 ± 2.9 Mg/ha (15.3 ± 1.3 ton/acre)] or large clumps [24.0 ± 2.4 Mg/ha (10.7 ± 1.0 ton/acre), $F_{2,8} = 7.7$, $p = 0.01$]. The biomass of large DWM within clumps after harvest [14.7 ± 2.5 Mg/ha (6.6 ± 1.1 ton/acre)] was similar to the volume within stands prior to harvest [12.4 ± 0.7 Mg/ha (5.5 ± 0.3 ton/acre); $F_{1,255} = 1.5$, $p = 0.23$].

The Grande Prairie area had a fairly even decay class distribution of DWM prior to harvest with a slight bias towards more advanced stages of decomposition (Fig. 4.1). Harvesting added a pulse of fresh DWM to the stands and appeared to reduce the amount of highly decomposed DWM (decay classes 5-7, Fig. 4.1).

Whitecourt Area
The post-harvest volume of large DWM was significantly lower than pre-harvest volume ($F_{1,456} = 5.9$, $p = 0.02$), however the volume of small DWM was twice as high after harvest than before ($F_{1,456} = 125.3$, $p < 0.001$, Table 4.3). The volume of all DWM was not significantly changed ($F_{1,456} = 0.5$, $p = 0.50$, Table 4.3). The volume of large DWM and total DWM was highest in stands with small clumps (large DWM, $F_{2,8} = 4.0$, $p = 0.06$; total DWM, $F_{2,8} = 4.8$, $p = 0.04$) ; however, the volume of small DWM was similar among all stand treatments ($F_{2,8} < 0.1$, $p = 0.99$, Table 4.4). The volume of small DWM was higher within clumps following harvest than within stands prior to harvest ($F_{1,255} = 4.9$, $p = 0.03$); however, there was no significant difference for large or total DWM (large DWM, $F_{1,255} = 1.2$, $p = 0.28$; total DWM, $F_{1,255} = 0.7$, $p = 0.40$, Table 4.3).

There was 31.4 Mg/ha (± 1.5 SE) [14.0 ton/acre (± 0.7 SE)] of large DWM prior to the harvest of stands in the Whitecourt area. This biomass was significantly higher after harvest [42.3 ± 1.9 Mg/ha (18.9 ± 0.8 ton/acre), $F_{1,456} = 25.5$, $p < 0.001$]. There was no significant difference in the biomass of large DWM between stand treatments in the Whitecourt area [small clumps, 48.7 ± 3.3 Mg/ha (21.7 ± 1.5 ton/acre); medium clumps, 42.9 ± 3.0 Mg/ha (19.1 ± 1.3 ton/acre); large clumps, 35.4 ± 3.3 Mg/ha (15.8 ± 1.5 ton/acre); $F_{2,8} = 2.0$, $p = 0.19$). The biomass of large DWM within clumps after harvest [35.5 ± 6.7 Mg/ha (15.8 ± 3.0 ton/acre)] was similar to the volume within stands prior to harvest [31.4 ± 1.5 Mg/ha (14.0 ± 0.7 ton/acre); $F_{1,255} = 0.4$, $p = 0.52$].

Most of the volume of DWM in the Whitecourt area was in advanced stages of decomposition prior to harvest (decay classes 6-7, Fig. 4.2). Decay classes 1-5 had an even distribution of DWM volume. Similar to the Grande Prairie area, harvesting added a pulse of fresh DWM to the stands and appeared to reduce the amount of highly decomposed DWM (decay classes 5-7, Fig. 4.2).

Pattern of DWM Decomposition
Grande Prairie Area - The DCA identified a decomposition gradient in aspen DWM (Axis 1 Eigen = 0.49) that was linearly correlated with decay class ($R^2 = 0.97$). The second axis extracted from the variables was weak in its explanatory value (Eigen = 0.05). Aspen logs appear to reside longer in decay classes 1, 5, and 7 than other decay classes (Fig. 4.3). Aspen logs lost 50% of their bark very early in decomposition (decay class 1); however the colonization of the logs by vegetation did not occur until very late in decomposition (50% cover in decay class 7). Aspen logs began fragmenting (wood state 3) in decay class 3 (Fig. 4.3).

The wood density of aspen logs decreased linearly as decomposition advanced (Fig. 4.3). Fresh aspen logs (decay class 1) had an average wood density of 0.411g/cm^3 (0.23 oz/in^3) while highly decomposed aspen logs (decay class 6) had an average wood density of 0.085g/cm^3 (0.005 oz/in^3). While wood density was decreasing, the moisture content of aspen logs increased from an average of 41.5% in fresh logs to 68.7% in highly decomposed logs (Fig. 4.3).

Whitecourt Area - The DCA identified a decomposition gradient in pine DWM (Axis 1 Eigen = 0.56) that was linearly correlated with decay class ($R^2 = 0.95$). The second axis

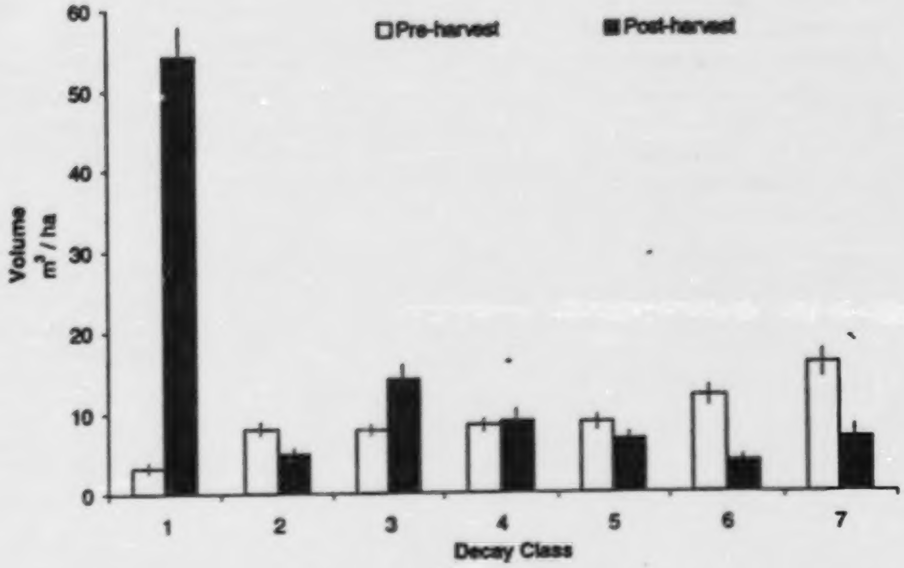

FIGURE 4.1. Average volume (+/- SE) of large DWM in different decay stages in the Grande Prairie area.

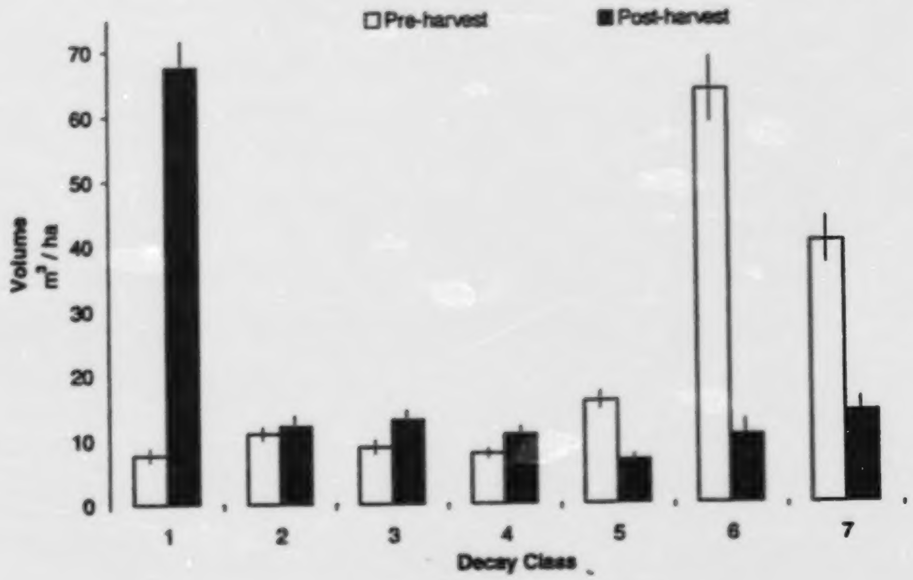

FIGURE 4.2. Average volume (+/- SE) of large DWM in different decay stages in the Whitecourt area.

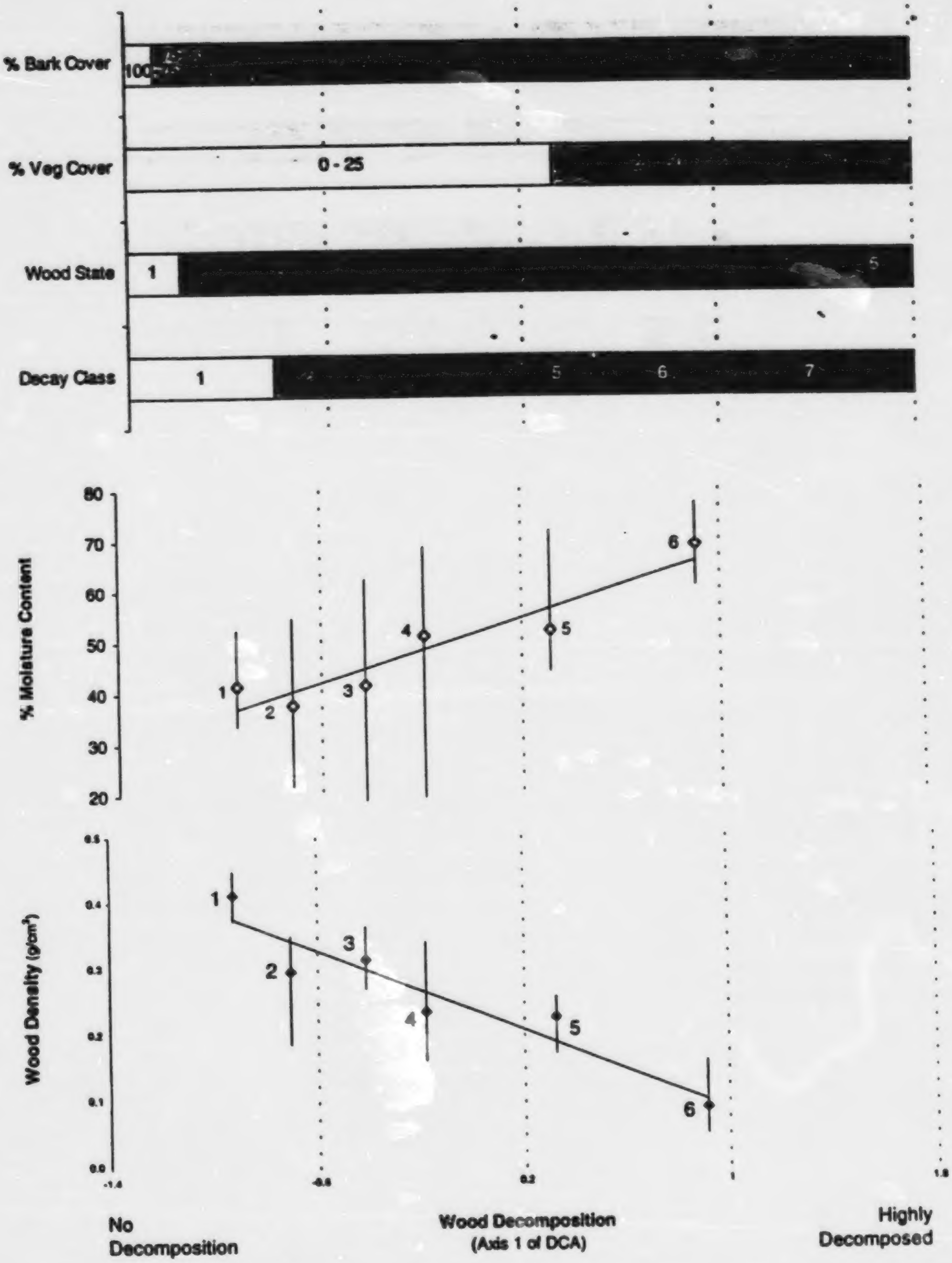

FIGURE 4.3. The progression of wood decomposition for aspen in the Grande Prairie study area. Ranges (error bars) for DWM in each of the decay classes (decay class numbers above the points) are shown for mean moisture content and mean wood density.

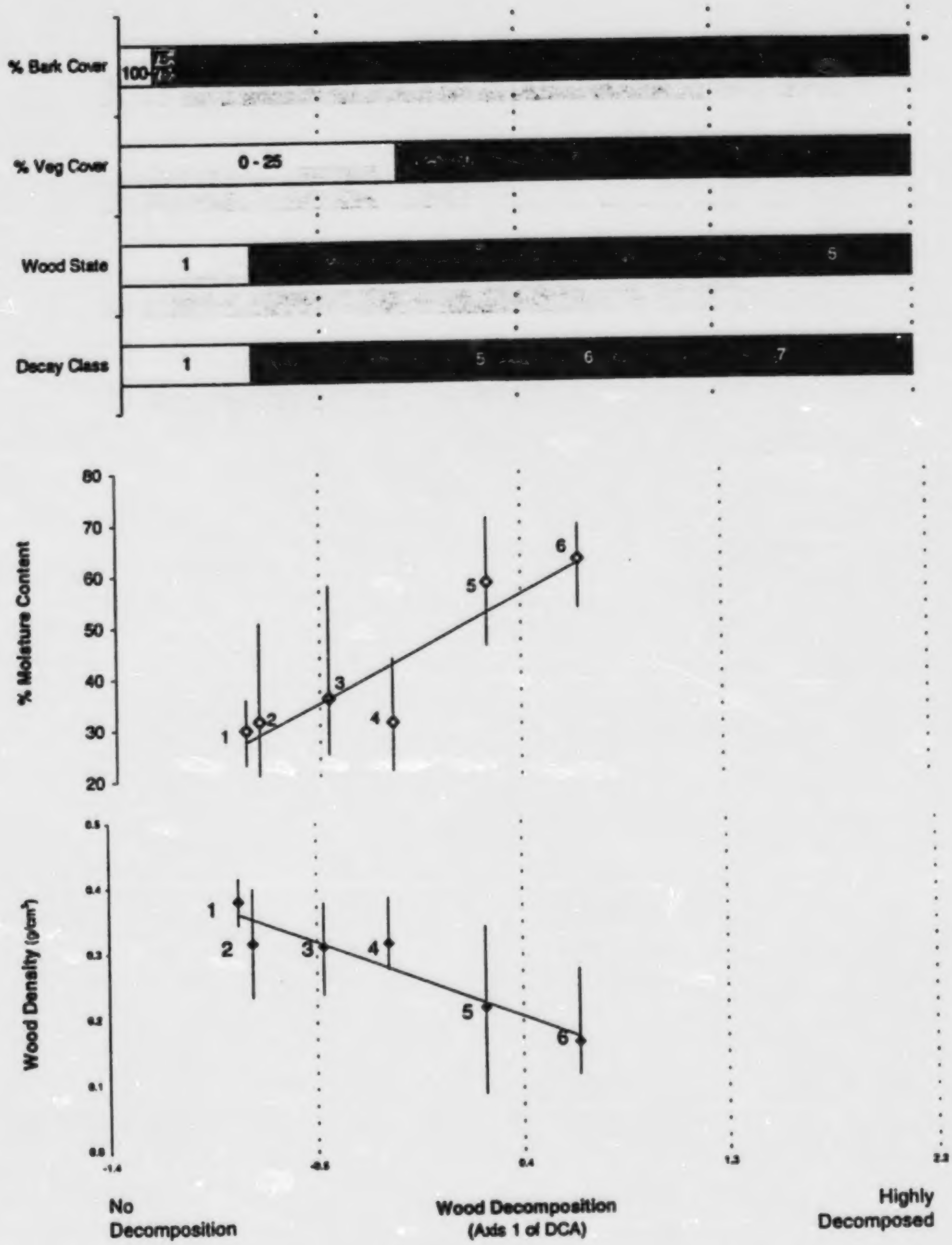

FIGURE 4.4. The progression of wood decomposition for lodgepole pine in the Whitecourt study area. Ranges (error bars) for DWM in each of the decay classes (decay class numbers above the points) are shown for mean moisture content and mean wood density.

extracted from the variables was weak in its explanatory value (Eigen = 0.02). Pine logs appear to spend more time in decay classes 1 and 7 than other decay classes (Fig. 4.4). Pine logs lost 50% of their bark very early in decomposition (decay class 1); however the colonization of the logs by vegetation did not occur until very late in decomposition (50% cover in decay class 5). Pine logs began fragmenting (wood state 3) in decay class 4 (Fig. 4.4).

The wood density of lodgepole pine logs decreased linearly as decomposition advanced (Figure 4.2). Newly fallen pine logs (decay class 1) had an average wood density of 0.379g/cm^3 (0.023 oz/in^3) while highly decomposed logs (decay class 6) had an average wood density of 0.160g/cm^3 (0.01 oz/in^3) (Fig. 4.4). At the same time the moisture content of pine logs increased from and average of 29.9% in newly fallen logs to 62.7% in highly decomposed logs (Fig. 4.4).

Discussion

Decomposition

Aspen DWM in the Grande Prairie area and lodgepole pine in Whitecourt area appear to decompose at the same rate when bark cover, wood density, and wood state are used as measures of decomposition. If previously published decomposition equations for spruce (k = 0.027, Foster and Lang 1982) are used to estimate ages for DWM in the Whitecourt area, decay classes 1-6 would have ages 0, 10, 11, 10, 30, and 45 years respectively. Similar equations for aspen (k = 0.07, Gosz 1980; see also Miller 1983) would suggest ages of 0, 5, 4, 8, 9, and 23 years for decay classes 1-6 in the Grande Prairie area. Interestingly, these predicted ages are highly correlated with the Axis 1 scores from the DCA (Grande Prairie, $R^2 = 0.88$; Whitecourt, $R^2 = 0.90$).

Pine-spruce DWM appears to be colonized by vegetation faster than does aspen DWM (Fig. 4.3 and 4.4). The shady, cool understory of the pine-spruce forest probably promotes moss cover (30 % ± 1.3 SE, Chapter 5) compared to the higher light levels in the understory of aspen forests (3.3% ± 0.3 SE moss cover). The rapid colonization of vegetation on pine-spruce DWM results in pine-spruce DWM spending almost 50% of its time in decay classes 6 and 7 (Fig. 4.3 and 4.4) whereas aspen DWM only spends a third of its time in these classes (Fig. 4.3 and 4.4). The rapid colonization of vegetation is correlated with a rapid uptake of moisture in pine-spruce DWM (Fig. 4.4).

Pre-harvest Characteristics

The pre-harvest decay class distribution of DWM in the Grande Prairie study area is very similar to previously published distributions of old (>120 years) aspen mixedwood forests in other areas of Alberta (Crites and Dale 1995; Lee et al. 1995, 1997). However, the volume [77.4 m^3/ha ± 3.4 SE (11.0 ton/acre ± 0.5 SE)] and biomass [12.4 Mg/ha ± 0.7 SE (5.53 ton/acre ± 0.3 SE)] of DWM in the Grande Prairie area was less than estimates from other aspen forests [Lac La Biche, 112.2 m^3/ha ± 8 SE (16.0 ton/acre ± 1.1 SE), Lee et al. 1995; 14.5 ± 2.3 Mg/ha (6.5 ± 1.0 ton/acre), Gosz 1980].

The current study is one of the first published on the characteristics of pine/spruce DWM. Decay class distributions of DWM from other pine-spruce dominated forest could not be found. The volume of pine-spruce DWM in the Whitecourt area [170.6 m^3/ha ± 7.9 SE (24.4 ton/acre ± 1.1 SE)] was only half of the volume in old growth Douglas-fir stands [396 m^3/ha ± 11 SE (56.6 ton/acre ± 1.6 SE), Sollins 1982]. In the Whitecourt area, there was 31.4 Mg/ha (± 1.5 SE) [14.0 ton/acre (± 0.7)]compared to 49 Mg/m^2 (45 ton/yd^2)in balsam fir forests (Lambert et al. 1980) and 81 Mg/ha (± 11 SE) [36 ton/acre (± 4.9 SE)] in old growth Douglas-fir stands (Sollins 1982). Decomposition rates of logs are dependent on size, species, and climate (Harmon 1982); therefore differences in the amounts of DWM between the current study and studies of other ecosystems are not surprising.

Effects of Harvest

Natural forest dynamics provide constant inputs of DWM (Harmon et al. 1986; Söderström 1988b) resulting in a relatively even distribution of size classes and decay stages. In this study, the pre-harvest data demonstrate a relatively even distribution of DWM among most decay stages. Abundant DWM in a variety of decay stages maximizes the amount and types of microhabitats, and therefore maximizes diversity of substrate for DWM-dependent taxa (Crites and Dale 1998). On the other hand, harvesting resulted in a pulse of DWM consisting mainly of small branches and twigs from felled and delimbed trees (see also McCarthy and Bailey 1994). As decomposition proceeds, pulses of DWM cause temporal heterogeneity in the abundance of species dependent on specific decay stages such as lichens and bryophytes (Söderström 1988b, Crites and Dale 1998) and of species dependent on specific sizes of DWM such as microarthropods (Abbott and Crossley 1982).

Unlike the Grande Prairie area, harvesting in the Whitecourt area reduced the amount of large DWM in advanced decay stages. The reduction is likely due to the compression or disintegration of highly decomposed logs by machinery (see also McCarthy and Bailey 1994) making them undetectable by observers. It is unlikely that these disturbed logs function within an ecosystem in the same way as large, undisturbed logs. A compacted log may not provide the amount, size, or quality of crevasses for animal shelter or seed germination. Larger logs have greater water-holding capacity and therefore offer better substrate for drought sensitive vegetation (Söderström 1988a). In addition, disintegrated logs have higher decomposition rates than intact logs, and therefore quickly release their nutrients and do not act as nutrient reservoirs for later in forest succession.

Management Implications

Stands that maximized the dispersion of residual material (small clumps) had higher volumes of DWM. This was probably due to two factors: 1) higher blow-down of trees within smaller clumps (Chapter 3), and 2) the decreased maneuverability of machinery when residual trees are highly dispersed. During harvest, operators commented on the difficulty in maneuvering around small clumps. This effect may have compacted less DWM and prevented 'clean' skidding of trees to roadsides.

Harvesting usually increases the amount of small, fresh DWM; however large DWM and DWM in advanced stages of decomposition is often reduced by harvesting. It may be possible to alter skidding patterns to maximize the amount of large DWM in harvested areas and to avoid the compaction and disturbance of naturally occurring DWM (McCarthy and Bailey 1994). By leaving residual trees in smaller clumps the amount of DWM left within a stand can be maximized. Furthermore, more abundant clumps may facilitate the dispersal of DWM-dependent taxa with limited dispersal capabilities, such as lichens and bryophytes.

Areas that fluctuate in humidity and temperature, such as clearcuts (Marra and Edmonds 1996), have slower decomposition of DWM and inhibit the colonization of vegetation (Muhle and LeBlanc 1975; Lesica et al. 1991; Gustafsson et al. 1992; Selva 1994). Larger residual clumps and stands with less edge:area ratios can be used to provide interior forest habitat and natural decomposition rates. Larger clumps are also more likely to maintain temporally homogenous compositions of species dependent on DWM. Only long-term studies will determine the appropriate size and distribution of clumps to retain vegetation sensitive to desiccation.

Future Research

Only long-term studies of DWM will determine which stand structure options will retain the biodiversity of DWM-dependent taxa on the landscape. Similar studies will also be able to determine the importance of DWM on ecosystem nutrient cycling.

DWM should be sampled again using similar methods within all stands at the following stages of forest succession:

> Canopy Closure (10-15 yr): At this point in succession, the initial pulse of small DWM generated from harvest will probably have decomposed and only larger DWM generated from harvest will remain (Lee and Crites in prep). This is also the stage when DWM-associated fungi may begin to colonize harvested areas (Amaranthus et al. (1994).

Mid Rotation (30-40 yr): Sunlight will be shaded from the forest floor and drought-sensitive vegetation may be able to begin colonizing DWM within the harvested areas. Furthermore, natural thinning will be adding DWM to the forest floor.

Mature Forest (60-80 yr): The maximum stage of DWM succession within a harvested landscape. Studies at this stage will determine whether or not particular DWM-dependent taxa can recover their distribution and abundance following harvest.

Literature Cited

Abbott, D.T. and Crossley, D.A., Jr. 1982. Woody litter decomposition following clear-cutting. Ecology 63: 35-42.

Amaranthus, M., Trappe, J.M., Bednar, L., and Arthur, D. 1994. Hypogeous fungal production in mature Douglas-fir forest fragments and surrounding plantations and its relation to coarse woody debris and animal mycophagy. Canadian Journal of Forest Research 24: 2157-2165.

Crites, S., and Dale, M. 1995. Relationships between nonvascular species and stand age and stand structure in aspen mixedwood forests in Alberta. In: Relationships between stand age, stand structure, and biodiversity in aspen mixedwood forests in Alberta. Jointly published by Alberta Environmental Centre (AECV95-R1), Vegreville, AB, and Canadian Forest Service (Project No. 0001A), Edmonton, AB. pp. 308.

Crites, S. and Dale, M.R.T. 1998. Diversity and abundance of bryophytes, lichens, and fungi in relation to woody substrate and successional stage in aspen mixedwood boreal forests. Canadian Journal of Botany 76: 641-651.

Foster, J.R., and Lang, G.E.. 1982. Decomposition of red spruce and balsam fir boles in the White Mountains of New Hampshire. Canadian Journal of Forest Research 12: 617-626.

Gosz, J.R. 1980. Biomass distribution and production budget for a nonaggrading forest ecosystem. Ecology 61: 507-514.

Gustafsson, L., Fiskesjö, A., Hallingbäck, T., Ingelög, T. 1992. Semi-natural deciduous broadleaved woods in southern Sweden: habitat factors of importance to some bryophyte species. Biological Conservation 59: 175-181.

Harmon, M.E., Franklin, J.F., Swanson, F.J., Sollins, P., Gregory, S.V., Lattin, J.D., Anderson, N.H., Cline, S.P., Aumen, N.G., Sedell, J.R., Lienkaemper, G.W., Cromack, K., Jr., and Cummins, K.W. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research 15: 133-302.

Lambert, R.L., Lang, G.E., and Reiners, W.A. 1980. Loss of mass and chemical change in decaying boles of a subalpine balsam fir forest. Ecology 61: 1460-1473.

Lee, P.C., Crites, S., Nietfeld, M., Van Nguyen, H., and Stelfox, J.B. 1995. Changes in snags and downed woody material characteristics in a chronosequence of aspen mixedwood forests in Alberta. *In* Relationships between stand age, stand structure, and biodiversity in aspen mixedwood forests in Alberta. *Edited by* J.B. Stelfox. Jointly published by Alberta Environment Centre (AECV95-R1), Vegreville, Alberta, and Canadian Forest Service (Project No. 0001A), Edmonton, Crites, S. and M. R. T. Dale. 1998. Diversity and abundance of bryophytes, lichens, and fungi in relation to woody substrate and successional stage in aspen mixedwood Alberta.

Lee, P.C., Crites, S., Nietfeld, M., Van Nguyen, H., and Stelfox, J.B. 1997. Characteristics and origins of deadwood material in aspen-dominated boreal forests. Ecological Applications 7: 691-701.

Lee, P., and Crites, S.. In Prep. Early successional deadwood dynamics in wildfire and harvest stands.

Lesica, P., McCune, B., Cooper, S.V., and Hong, W.S. 1991. Differences in lichen and bryophyte communities between old-growth and managed second-growth forests in the Swan Valley, Montana. *Canadian Journal of* Botany 69: 1745-1755.

MacMillan, P.C. 1981. Log decomposition in Donaldson's Woods, Spring Mill State Park, Indiana. The American Midland Naturalist 106: 335-344.

Marra, J.L., and Edmonds, R.L. 1996. Coarse woody debris and soil respiration in a clearcut on the Olympic Peninsula, Washington, U.S.A. Canadian Journal of Forest Research 26: 1337-1345.

McCarthy, B.C., and R.R. Bailey. 1994. Distribution and abundance of coarse woody debris in a managed forest landscape of the central Appalachians. Canadian Journal of Forest Research 24: 1317-1329.

McCullough, H. A. 1948. Plant succession on fallen logs in a virgin spruce-fir forest. Ecology 29: 508-513.

Miller, W.E. 1983. Decomposition rates of aspen bole and branch litter. Forest Science 29(2): 351-356.

Muhle, H., and F. LeBlanc. 1975. Bryophyte and lichen succession on decaying logs: I. Analysis along an evaporational gradient in eastern Canada. Journal of Hattori Botanical Laboratory 39: 1-33.

Selva, S.B. 1994. Lichen diversity and stand continuity in the northern hardwoods and spruce-fir forests of northern New England and western New Brunswick. The Bryologist 97(4): 424-429.

Söderström, L. 1988a. Sequence of bryophytes and lichens in relation to substrate variables of decaying coniferous wood in northern Sweden. Nordic Journal of Botany 8: 89-97.

Söderström, L. 1988b. The occurrence of epixylic bryophyte and lichen species in an old natural and a managed forest stand in northeast Sweden. Biological Conservation 45: 169-178.

Sollins, P. 1982. Input and decay of coarse woody debris in coniferous stands in western *Oregon and Washington.* Canadian Journal of Forest Research 12: 18-28.

SPSS Inc. 1997. SPSS 7.5.1 for Windows.

Thompson, J. N. 1980. Treefalls and colonization patterns of temperate forest herbs. American Midland Naturalist 104: 176-184.

Van Wagner, C.E. 1968. 'The line intersect method in forest fuel sampling. Forest Science 14: 20-26.

Zar, J. H. 1984. Biostatistical Analysis, 2nd Edition. Prentice Hall, New Jersey.

CHAPTER 5. A SUMMARY OF UNDERSTORY VEGETATION IN THE BOREAL FOREST PRIOR TO HARVEST.

Christine Gray and Steve Bradbury

Introduction

Due to funding constraints, only preliminary surveys were conducted for the understory vegetation during the 1997 field season. This report is a summary of that data. More detailed data collection, summary, and analyses will be conducted during 1999/2000.

Methods

In both study areas, vegetation was sampled at half of the 1 ha (2.47 acre) sites (every second site) within each stand. Each 1 ha site was divided into four equal quadrants (Fig 2.3). Two herb plots [0.5 x 0.5 m (1.6 x 1.6 ft.)] were systematically placed in all four quadrants (8 herb plots per site and 40 herb plots per stand).

For sampling purposes, vegetation was divided into two height strata. The low understory stratum included all herbs regardless of height, and trees and shrubs that were <0.5 m (<1.6 ft.) in height. The high understory stratum included all shrubs ≥0.5 m (≥1.6 ft.) in height, and trees that were between 0.5 and 1.5 m (1.6–4.9 ft.) tall. Shrubs were divided into height classes of **0** = 0.5-1.39 m (1.6-4.6 ft.); **1** = 1.4-2.9 m (4.7-9.5 ft.); **2** = 3.0-4.9 m (9.6-16.1 ft.); **3** = 5.0-9.9 m (16.2-32.5 ft.); **4** = 10.0-19.9 m (32.6-65.6 ft.); and **5** = ≥20.0 m (≥ 65.6 ft.). In the Grande Prairie study area one high understory plot [2.0 x 2.0 m (6.6 x 6.6 ft.)] was systematically placed in two of the four randomly selected quadrants (2 plots per site and 10 plots per stand). In the Whitecourt study area one high understory plot was placed in all four quadrants (4 plots per site and 20 plots per stand). Low understory plots were nested within high understory plots. Sampling was conducted from mid June through mid August 1997.

Percent of ground surface cover was visually estimated for litter, wood, moss, lichen, and grass, and percent of ground surface covered by a vertical projection of leaves and stems was estimated for individual species of herbs, shrubs, and trees within each plot. In the Whitecourt study area percent of the ground surface cover by individual moss species was also recorded as they were the dominant ground cover. Grasses and lichens were not identified to species at either study area. Unknown species were collected and later identified at the University of Alberta herbarium. Nomenclature follows Moss (1992) for vascular species. For nonvascular species, nomenclature follows Vitt et al. (1988).

Results

Grande Prairie Study Area

Eighty-three species were found in the low understory strata, of which 55 were herbs, 25 were shrubs, and three were tree species (Table 5.1). Of the 25 shrub species found in the low understory strata, 9 were not found in the high understory strata. Within the low understory strata, much of the ground was covered by litter (43% cover) although herbs and shrubs were also abundant (39% and 18% cover respectively). Percent cover of grasses, mosses, and lichen were much lower in this strata (Table 5.1). Bunchberry was the most abundant herb (6% cover), with Prickly rose and Bracted honeysuckle the most abundant shrubs (5% cover for each) in the low understory strata. Trees had low percent cover in both the low and high understory strata (Table 5.1).

The high understory strata contained 21 species, of which 18 were shrubs and 3 were tree species (Table 5.1). Of these, River alder and Common Labrador tea were unique to the high understory stratum. The most abundant shrubs in the high understory stratum were Bracted honeysuckle (6% cover), Canadian buffaloberry (5% cover), and Prickly rose (5% cover). Trembling aspen was the most abundant tree species (1% cover) in the high understory stratum.

Whitecourt Study Area

Seventy-six vascular species (including 53 herbs, 19 shrubs, and 4 tree species) and 22

TABLE 5.1. Mean percent cover (± SE) of understory vegetation in the Grande Prairie study area during 1997. Individual species are grouped by strata level and family.

Common Name	Species Name	Species Code	Mean Percent Cover
Litter Total Cover			**43.17 ± 0.92**
Wood Total Cover			**5.31 ± 0.45**
Lichen Total Cover			**0.01 ± 0.00**
Moss Total Cover			**3.27 ± 0.31**
Grass Total Cover			**4.81 ± 0.26**
Herb Total Cover (low understory < 0.5 m)			**38.92 ± 1.01**
Lycopodiaciaceae			
Stiff club-moss	*Lycopodium annotinum*	LYC ANN	0.06 ± 0.02
Ground-cedar	*Lycopodium complanatum*	LYC COM	0.04 ± 0.04
Equisetaceae			
Common horsetail	*Equisetum arvense*	EQU ARV	0.23 ± 0.04
Woodland horsetail	*Equisetum sylvaticum*	EQU SYL	0.04 ± 0.01
Polypodiaceae			
Lady fern	*Athyrium filix-femina*	ATH FIL	0.01 ± 0.01
Oak fern	*Gymnocarpium dryopteris*	GYM DRY	0.01 ± 0.01
unknown fern		UNK FER	0.08 ± 0.08[1]
Liliaceae			
Fairybells	*Disporum trachycarpum*	DIS TRA	0.01 ± 0.01
Wild lily-of-the-valley	*Maianthemum canadense*	MAI CAN	0.67 ± 0.08
Star-flowered false solomon's-seal	*Smilacina stellata*	SMI STE	0.04 ± 0.03
Three-leaved false solomon's-seal	*Smilacina trifolia*	SMI TRI	0.02 ± 0.01
unknown lily	*Smilacina* spp.	SMI SPP	0.02 ± 0.02
Orchidaceae			
Rattlesnake-plantain	*Goodyera oblongifolia*	GOO OBL	0.01 ± 0.01
Blunt-leaved bog-orchid	*Habenaria obtusata*	HAB OBT	0.01 ± 0.01
Ranunculaceae			
Red and white baneberry	*Actaea rubra*	ACT RUB	0.32 ± 0.06
Veiny meadow rue	*Thalictrum venulosum*	THA VEN	0.01 ± 0.01
Saxifragaceae			
Bishop's cap	*Mitella nuda*	MIT NUD	0.78 ± 0.08
Rosaceae			
Wild strawberry	*Fragaria virginiana*	FRA VIR	2.47 ± 0.13
Dewberry	*Rubus pubescens*	RUB PUB	2.17 ± 0.15
Leguminosae			
Creamy peavine	*Lathyrus ochroleucus*	LAT OCH	3.01 ± 0.18
American vetch	*Vicia americana*	VIC AME	1.20 ± 0.11
Violaceae			
Canada violet	*Viola canadensis*	VIO CAN	0.26 ± 0.07
Kidney-leaved violet	*Viola renifolia*	VIO REN	0.53 ± 0.07
Onagraceae			
Fireweed	*Epilobium angustifolium*	EPI ANG	2.65 ± 0.19

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[1] Sum of two unidentified fern species.

TABLE 5.1 cont.

Common Name	Species Name	Species Code	Mean Percent Cover
Araliaceae			
Wild sarsaparilla	*Aralia nudicaulis*	ARA NUD	1.83 ± 0.25
Umbelliferae			
Cow-parsnip	*Heracleum lanatum*	HER LAN	0.23 ± 0.09
Cornaceae			
Bunchberry	*Cornus canadensis*	COR CAN	6.11 ± 0.32
Pyrolaceae			
One-sided wintergreen	*Orthilia secunda*	ORT SEC	0.04 ± 0.03
Common pink wintergreen	*Pyrola asarifolia*	PYR ASA	0.94 ± 0.10
Northern starflower	*Trientalis borealis*	TRI BOR	0.02 ± 0.01
Gentianaceae			
Spurred gentian	*Halenia deflexa*	HAL DEF	0.06 ± 0.02
Boraginaceae			
Tall lungwort	*Mertensia paniculata*	MER PAN	1.61 ± 0.17
Labiatae			
Marsh skullcap	*Scutellaria galericulata*	SCU GAL	0.01 ± 0.01
Scrophulariaceae			
Common red paintbrush	*Castilleja miniata*	CAS MIN	0.52 ± 0.09
Rubiaceae			
Northern bedstraw	*Galium boreale*	GAL BOR	1.36 ± 0.09
Sweet-scented bedstraw	*Galium triflorum*	GAL TRI	0.06 ± 0.01
Caprifoliaceae			
Twinflower	*Linnaea borealis*	LIN BOR	2.89 ± 0.17
Compositae			
Common yarrow	*Achillia millefolium*	ACH MIL	0.26 ± 0.04
Heart-leaved arnica	*Arnica cordifolia*	ARN COR	0.36 ± 0.07
Lindley's aster	*Aster ciliolatus*	AST CIL	0.90 ± 0.13
Showy aster	*Aster conspicuus*	AST CON	3.37 ± 0.29
Purple-stemmed aster	*Aster puniceus*	AST PUN	0.03 ± 0.02
Narrow-leaved hawkweed	*Hieracium umbellatum*	HIE UMB	0.96 ± 0.17
Palmate-leaved coltsfoot	*Petasites palmatus*	PET PAL	2.38 ± 0.16
Arrow-leaved coltsfoot	*Petasites sagittatus*	PET SAG	0.14 ± 0.08
Canadian butterweed	*Senecio pauperculus*	SEN PAU	0.01 ± 0.01
Canada goldenrod	*Solidago canadensis*	SOL CAN	0.09 ± 0.04
Late goldenrod	*Solidago gigantea*	SOL GIG	0.03 ± 0.02
Common dandelion	*Taraxacum officinale*	TAR OFF	0.01 ± 0.01
unknowns		UNK	0.09 ± 0.07[2]
Shrub Total Cover (low understory < 0.5 m)			**18.21 ± 0.55**
Salicaceae			
willow	*Salix* spp.	SAL SPP	0.10 ± 0.04
Betulaceae			
Green alder	*Alnus crispa*	ALN CRI	0.03 ± 0.02
Beaked hazelnut	*Corylus cornuta*	COR COR	0.07 ± 0.05
Grossulariaceae			
Skunk currant	*Ribes glandulosum*	RIB GLA	0.01 ± 0.01
Northern black currant	*Ribes hudsonianum*	RIB HUD	0.06 ± 0.03
Black gooseberry	*Ribes lacustre*	RIB LAC	0.04 ± 0.02

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[2] Sum of five unidentified herb species.

TABLE 5.1 cont.

Common Name	Species Name	Species Code	Mean Percent Cover
Northern gooseberry	*Ribes oxyacanthoides*	RIB OXY	0.16± 0.03
Wild red currant	*Ribes triste*	RIB TRI	0.36 ± 0.08
unknown currant	*Ribes spp*	RIB SPP	0.10 ± 0.04
Rosaceae			
Saskatoon	*Amelanchier alnifolia*	AME ALN	0.29 ± 0.08
Pin cherry	*Prunus pensylvanica*	PRU PEN	0.03 ± 0.02
Choke cherry	*Prunus virginiana*	PRU VIR	0.01 ± 0.01
Prickly rose	*Rosa acicularis*	ROS ACI	5.33 ± 0.24
Wild red raspberry	*Rubus idaeus*	RUB IDA	0.70 ± 0.10
Narrow-leaved meadowsweet	*Spiraea alba*	SPI BET	1.09 ± 0.13
Elaegnaceae			
Canada buffaloberry	*Shepherdia canadensis*	SHE CAN	0.62 ± 0.09
Cornaceae			
Red-osier dogwood	*Cornus stolonifera*	COR STO	0.12 ± 0.05
Ericaceae			
Common bearberry	*Arctostaphylos uva-ursi*	ARC UVA	0.01 ± 0.01
Common labrador tea	*Ledum groenlandicum*	LED GRO	0.05 ± 0.03
Velvet-leaved blueberry	*Vaccinium myrtilloides*	VAC MYR	0.78 ± 0.14
Bog cranberry	*Vaccinium vitis-idaea*	VAC VIT	0.04 ± 0.02
Caprifoliaceae			
Twining honeysuckle	*Lonicera dioica*	LON DIO	0.28 ± 0.06
Bracted honeysuckle	*Lonicera involucrata*	LON INV	5.22 ± 0.36
Common snowberry	*Symphoricarpos albus*	SYM ALB	0.04 ± 0.02
Low bush-cranberry	*Viburnum edule*	VIB EDU	2.67 ± 0.29
Shrub Total Cover (high understory > 0.5 m)			29.36 ± 1.95
Salicaceae			
willow	*Salix* spp.	SAL SPP	3.48 ± 0.51
Betulaceae			
Green alder	*Alnus crispa*	ALN CRI	1.64 ± 0.62
River alder	*Alnus tenuifolia*	ALN TEN	1.43 ± 0.83
Beaked hazelnut	*Corylus cornuta*	COR COR	0.18 ± 0.13
Grossulariaceae			
Northern gooseberry	*Ribes oxyacanthoides*	RIB OXY	0.07 ± 0.04
unknown currant	*Ribes* spp.	RIB SPP	0.02 ± 0.01
Rosaceae			
Saskatoon	*Amelanchier alnifolia*	AME ALN	0.94 ± 0.27
Pin cherry	*Prunus pensylvanica*	PRU PEN	0.41 ± 0.16
unknown cherry	*Prunus* spp.	PRU SPP	0.01 ± 0.01
Choke cherry	*Prunus virginiana*	PRU VIR	0.02 ± 0.01
Prickly rose	*Rosa acicularis*	ROS ACI	5.02 ± 0.58
Wild red raspberry	*Rubus idaeus*	RUB IDA	0.14 ± 0.07
Elaegnaceae			
Canada buffaloberry	*Shepherdia canadensis*	SHE CAN	5.24 ± 1.01
Cornaceae			
Red-osier dogwood	*Cornus stolonifera*	COR STO	0.05 ± 0.04
Ericaceae			
Common labrador tea	*Ledum groenlandicum*	LED GRO	0.08 ± 0.08

TABLE 5.1 cont.

Common Name	Species Name	Species Code	Mean Percent Cover
Caprifoliaceae			
Twining honeysuckle	*Lonicera dioica*	LON DIO	0.09 ± 0.06
Bracted honeysuckle	*Lonicera involucrata*	LON INV	6.27 ± 1.14
Low bush-cranberry	*Viburnum edule*	VIB EDU	4.28 ± 0.75
Tree Total Cover (low understory < 0.5 m)			**0.07 ± 0.04**
Pinaceae			
White spruce	*Picea glauca*	PIC GLA	0.01 ± 0.01
Salicaceae			
Trembling aspen	*Populus tremuloides*	POP TRE	0.05 ± 0.04
Betulaceae			
Paper birch	*Betula papyrifera*	BET PAP	0.01 ± 0.01
Tree Total Cover (high understory 0.5 – 1.5 m)			**0.87 ± 0.31**
Pinaceae			
White spruce	*Picea glauca*	PIC GLA	0.04 ± 0.04
Salicaceae			
Trembling aspen	*Populus tremuloides*	POP TRE	0.74 ± 0.28
Betulaceae			
Paper birch	*Betula papyrifera*	BET PAP	0.08 ± 0.08

TABLE 5.2. Mean percent cover (± SE) of understory vegetation in the Whitecourt study area during 1997. Individual species are grouped by strata level and family.

Common Name	Species Name	Species Code	Mean Percent Cover
Litter Total Cover			**29.56 ± 1.08**
Wood Total Cover			**4.76 ± 0.36**
Lichen Total Cover			**0.08 ± 0.04**
club lichen	*Cladonia* spp.	CLA SPP	0.01 ± 0.01
Studded leather lichen	*Peltigera aphthosa*	PEL APH	0.01 ± 0.01
Finger felt lichen	*Peligera neopolydactyla*	PEL NEO	0.04 ± 0.03
Moss Total Cover			**30.61 ± 1.28**
sphagnum	*Sphagnum* spp.	SPH SPP	3.74 ± 0.68
Common hair-cap	*Polytrichum commune*	POL COM	0.39 ± 0.12
Juniper hair-cap	*Polytrichum juniperinum*	POL JUN	0.30 ± 0.08
Slender hair-cap	*Polytrichum strictum*	POL STR	0.49 ± 0.12
Common northern lantern moss	*Cinclidium stygium*	CIN STY	1.46 ± 0.20
Knight's plume	*Ptilium crista-castrensis*	PTI CRI	17.46 ± 1.03
Big red stem	*Pleurozium schreberi*	PLE SCH	14.75 ± 0.91
Stair-step moss	*Hylocomium splendens*	HYL SPL	0.80 ± 0.14
Broom moss	*Dicranum scoparium*	DIC SCO	0.27 ± 0.06

TABLE 5.2 cont

Common Name	Species Name	Species Code	Mean Percent Cover
unknown mosses		MOSS	0.56 ± 0.39[3]
Grass Total Cover			**3.10 ± 0.36**
Herb Total Cover (low understory < 0.5 m)			**28.24 ± 0.95**
Lycopodiaceae			
Stiff club-moss	*Lycopodium annotinum*	LYC ANN	2.61 ± 0.21
Ground-cedar	*Lycopodium complanatum*	LYC COM	0.03 ± 0.02
Equisetaceae			
Common horsetail	*Equisetum arvense*	EQU ARV	0.52 ± 0.08
Woodland horsetail	*Equisetum sylvaticum*	EQU SYL	0.76 ± 0.13
Polypodiaceae			
Lady fern	*Athyrium filix-femina*	ATH FIL	1.91 ± 0.49
Oak fern	*Gymnocarpium dryopteris*	GYM DRY	2.53 ± 0.21
Cyperaceae			
unknown sedge	*Carex* spp.	SED SPP	0.01 ± 0.01
Liliaceae			
Fairybells	*Disporum trachycarpum*	DIS TRA	0.48 ± 0.08
Wild lily-of-the-valley	*Maianthemum canadense*	MAI CAN	0.22 ± 0.04
False solomon's-seal	*Smilacina racemosa*	SMI RAC	0.01 ± 0.01
Star-flowered false solomon's-seal	*Smilacina stellata*	SMI STE	0.03 ± 0.02
Three-leaved false solomon's-seal	*Smilacina trifolia*	SMI TRI	0.05 ± 0.02
unknown lily	*Smilacina* spp.	SMI SPP	0.02 ± 0.01
Indian hellebore	*Veratrum viride*	VER VIR	0.04 ± 0.03
Orchidaceae			
Heart-leaved twayblade	*Listera cordata*	LIS COR	0.01 ± 0.00
Urticaceae			
Stinging nettle	*Urtica dioica*	URT DIO	0.05 ± 0.03
Ranunculaceae			
Red and white baneberry	*Actaea rubra*	ACT RUB	0.05 ± 0.03
Goldthread	*Coptis trifolia*	COP TRI	0.16 ± 0.04
Tall larkspur	*Delphinium glaucum*	DEL GLA	0.01 ± 0.01
Veiny meadow rue	*Thalictrum venulosum*	THA VEN	0.03 ± 0.01
Saxifragaceae			
Bishop's cap	*Mitella nuda*	MIT NUD	0.58 ± 0.07
Three-leaved foamflower	*Tiarella trifoliata*	TIA TRI	1.38 ± 0.16
Rosaceae			
Wild strawberry	*Fragaria virginiana*	FRA VIR	0.02 ± 0.01
Purple avens	*Geum rivale*	GEU RIV	0.02 ± 0.01
Cloudberry	*Rubus chamaemorus*	RUB CHA	0.30 ± 0.09
Five-leaved bramble	*Rubus pedatus*	RUB PED	2.90 ± 0.22
Dewberry	*Rubus pubescens*	RUB PUB	1.91 ± 0.15
Leguminosae			
Creamy peavine	*Lathyrus ochroleucus*	LAT OCH	0.01 ± 0.01
Balsaminaceae			
Spotted touch-me-not	*Impatiens capensis*	IMP CAP	0.01 ± 0.01
Violaceae			
Kidney-leaved violet	*Viola renifolia*	VIO REN	0.19 ± 0.04

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[3] Sum of ten unidentified moss species.

TABLE 5.2 cont.

Common Name	Species Name	Species Code	Mean Percent Cover
Onagraceae			
Small enchanter's nightshade	*Circaea alpina*	CIR ALP	0.01 ± 0.01
Fireweed	*Epilobium angustifolium*	EPI ANG	0.93 ± 0.11
Araliaceae			
Wild sarsaparilla	*Aralia nudicaulis*	ARA NUD	2.36 ± 0.25
Umbelliferae			
Cow-parsnip	*Heracleum lanatum*	HER LAN	0.06 ± 0.06
Cornaceae			
Bunchberry	*Cornus canadensis*	COR CAN	4.34 ± 0.20
Pyrolaceae			
One-sided wintergreen	*Orthilia secunda*	ORT SEC	0.01 ± 0.00
Common pink wintergreen	*Pyrola asarifolia*	PYR ASA	0.29 ± 0.06
Green wintergreen	*Pyrola virens*	PYR VIR	0.05 ± 0.02
Boraginaceae			
Tall lungwort	*Mertensia paniculata*	MER PAN	0.29 ± 0.07
Scrophulariaceae			
Common red paintbrush	*Castilleja miniata*	CAS MIN	0.01 ± 0.01
Rubiaceae			
Northern bedstraw	*Galium boreale*	GAL BOR	0.01 ± 0.00
Sweet-scented bedstraw	*Galium triflorum*	GAL TRI	0.06 ± 0.02
Caprifoliaceae			
Twinflower	*Linnaea borealis*	LIN BOR	1.94 ± 0.15
Compositae			
Heart-leaved arnica	*Arnica cordifolia*	ARN COR	0.29 ± 0.06
Narrow-leaved hawkweed	*Hieracium umbellatum*	HIE UMB	0.03 ± 0.03
Palmate-leaved coltsfoot	*Petasites palmatus*	PET PAL	0.65 ± 0.08
Arrow-leaved groundsel	*Senecio triangularis*	SEN TRI	0.01 ± 0.01
Late goldenrod	*Solidago gigantea*	SOL GIG	0.03 ± 0.03
Common dandelion	*Taraxacum officinale*	TAR OFF	0.01 ± 0.01
unknowns		UNK	0.06 ± 0.06[4]
Shrub Total Cover (low understory < 0.5 m)			**10.63 ± 0.52**
Salicaceae			
willow	*Salix* spp.	SAL SPP	0.01 ± 0.01
Betulaceae			
Green alder	*Alnus crispa*	ALN CRI	0.04 ± 0.02
River alder	*Alnus tenuifolia*	ALN TEN	0.01 ± 0.01
Grossulariaceae			
Northern black currant	*Ribes hudsonianum*	RIB HUD	0.02 ± 0.02
Northern gooseberry	*Ribes oxyacanthoides*	RIB OXY	0.47 ± 0.09
Wild red currant	*Ribes triste*	RIB TRI	0.11 ± 0.04
unknown currant	*Ribes* spp.	RIB SPP	0.06 ± 0.02
Rosaceae			
Saskatoon	*Amelanchier alnifolia*	AME ALN	0.09 ± 0.04
Prickly rose	*Rosa asicularis*	ROS ACI	0.95 ± 0.11
Wild red raspberry	*Rubus idaeus*	RUB IDA	0.33 ± 0.09
Western mountain ash	*Sorbus scopulina*	SOR SCO	0.01 ± 0.01
Araliaceae			
Devil's club	*Oplopanax horridum*	OPL HOR	0.91 ± 0.26

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[4] Sum of four identified herb species.

TABLE 5.2 cont.

Common Name	Species Name	Species Code	Mean Percent Cover
Ericeaceae			
Creeping snowberry	*Gaultheria hispidula*	GAU HIS	0.56 ± 0.09
Common Labrador tea	*Ledum groenlandicum*	LED GRO	2.55 ± 0.28
Black huckleberry	*Vaccinium membranaceum*	VAC MEM	0.16 ± 0.05
Velvet-leaved blueberry	*Vaccinium myrtilloides*	VAC MYR	2.12 ± 0.21
Bog cranberry	*Vaccinium vitis-idaea*	VAC VIT	0.92 ± 0.12
Caprifoliaceae			
Bracted honeysuckle	*Lonicera involucrata*	LON INV	0.60 ± 0.1
Low bush-cranberry	*Viburnum edule*	VIB EDU	0.72 ± 0.11
Shrub Total Cover (high understory > 0.5 m)			**7.00 ± 0.72**
Salicaceae			
willow	*Salix* spp.	SAL SPP	0.32 ± 0.09
Betulaceae			
Green alder	*Alnus crispa*	ALN CRI	1.05 ± 0.34
River alder	*Alnus tenuifolia*	ALN TEN	0.17 ± 0.11
Grossulariaceae			
Northern black currant	*Ribes hudsonianum*	RIB HUD	0.04 ± 0.04
Northern gooseberry	*Ribes oxyacanthoides*	RIB OXY	0.14 ± 0.06
Wild red currant	*Ribes triste*	RIB TRI	0.01 ± 0.01
unknown currant	*Ribes* spp.	RIB SPP	0.03 ± 0.02
Rosaceae			
Saskatoon	*Amelanchier alnifolia*	AME ALN	0.01 ± 0.01
Prickly rose	*Rosa acicularis*	ROS ACI	1.20 ± 0.18
Wild red raspberry	*Rubus idaeus*	RUB IDA	0.16 ± 0.13
Araliaceae			
Devil's club	*Oplopanax horridum*	OPL HOR	1.57 ± 0.44
Ericaceae			
Common Labrador tea	*Ledum groenlandicum*	LED GRO	1.06 ± 0.24
Caprifoliaceae			
Bracted honeysuckle	*Lonicera involucrata*	LON INV	0.66 ± 0.19
Low bush-cranberry	*Viburnum edule*	VIB EDU	0.59 ± 0.13
Tree Total Cover (low understory < 0.5 m)			**0.49 ± 0.10**
Pinaceae			
Balsam fir	*Abies balsamea*	ABI BAL	0.40 ± 0.09
White spruce	*Picea glauca*	PIC GLA	0.01 ± 0.01
Black spruce	*Picea mariana*	PIC MAR	0.02 ± 0.01
Betulaceae			
Paper birch	*Betula papyrifera*	BET PAP	0.07 ± 0.03
Tree Total Cover (high understory 0.5 – 1.5 m)			**1.78 ± 0.27**
Pinaceae			
Balsam fir	*Abies balsamea*	ABI BAL	1.03 ± 0.23
White spruce	*Picea glauca*	PIC GLA	0.01 ± 0.01
Black spruce	*Picea mariana*	PIC MAR	0.16 ± 0.06
Salicaceae			
Balsam poplar	*Populus balsamifera*	POP BAL	0.02 ± 0.02
Trembling aspen	*Populus tremuloides*	POP TRE	0.03 ± 0.02
Betulaceae			
Paper birch	*Betula papyrifera*	BET PAP	0.53 ± 0.13

nonvascular species (19 mosses and 3 lichens) were found in the low understory strata (Table 5.2). Within the low understory strata moss, litter and herbs were equally abundant (31%, 30% and 28% cover respectively). Knight's plume (17% cover) and Big red stem (15% cover) were the most abundant moss species. Bunchberry was the most abundant herb (4% cover) with Common Labrador tea (3% cover) and Velvet-leaved blueberry (2% cover) the most abundant shrubs in the low understory strata. Balsam fir was the most abundant tree species in both the low and high understory strata (Table 5.2).

The high understory strata contained a total of 20 species, of which 14 were shrubs and 6 were trees (Table 5.2). Five shrub species found in the low understory strata were not found in the high understory strata. Devil's club (2% cover) was the most abundant shrub species in the high understory strata. Balsam poplar and Trembling aspen were the only tree species found in the high understory strata and not the low understory strata.

Discussion

The Grande Prairie study area had a well developed low and high understory strata; shrubs were tall and abundant throughout the study area and the herbs covered a majority of the ground space. This abundance may have been due to the deciduous canopy allowing light to penetrate through to the understory.

The Whitecourt study area had a high density of coniferous trees (Chapter 4). The low and high understory strata were less developed in this area, possibly due to shading. An abundant moss community was present, possibly as a result of the shaded and damp conditions.

Due to funding constraints only preliminary surveys and summary were conducted for the preharvest understory vegetation.

Future research

Surveys and analysis of post-harvest vegetation will take place during 1999/2000. Herbs and shrubs will be sampled using square plots of the dimensions described in the Methods section of this chapter. Approximately half of the herb plots will also be surveyed for shrubs, and in these cases, herb plots will be nested within the larger shrub plots. In each of the areas with small clumps (Figure 5.1), four clumps will be sampled for herbs and shrubs (4 stands per area x 4 clumps per stand = 16 clumps per area). In each of the stands with small clumps, one herb and one shrub plot will be located at the centre of the clump, and at the clump edge in each ordinal direction. At two of the clumps within each stand, additional herb plots will be located at 10 m (11 yd.) and 20 m (22 yd.) out from the clump edge in each ordinal direction. Shrub plots will be surveyed only at 20 m (22 yd.) out from the clump edge.

In stands with medium clumps (Figure 5.2), two clumps will be sampled for herbs and shrubs (4 stands per area x 2 clumps per stand = 8 clumps per area). Herb plots will be located at the clump centre and at 10 m (11 yd.) and 20 m (22 yd.) intervals in each ordinal direction to the clump edge. Two additional herb plots will be located at 10 m (11 yd.) and 20 m (22 yd.) intervals from the clump edge into the cutover area in each ordinal direction. Approximately half of the herb plots will also have shrub plots. Shrub plots will be located in the centre of the clump, at the clump edge, and at the 20 m (22 yd.) interval in the cutover area in each ordinal direction.

In stands with large clumps (Figure 5.3), one clump will be sampled for herbs and shrubs (4 stands per area x 1 clump per stand = 4 clumps per area). Herb plots will be located at the centre of the clump, and at 10 m (11 yd.), 20 m (22 yd.), 30 m (33 yd.), and 40 m (44 yd.) intervals in each ordinal direction to the clump edge. Two additional herb plots will be located at 10 m (11 yd.) and 20 m (22 yd.) intervals into the cutover area in each ordinal direction from the clump edge. Again, approximately half of the herb plots will be surveyed for shrubs. Shrub plots will be located in the centre of the clump, at the 20 m (22 yd.) interval within the clump, at the clump edge, and at the 20 m (22 yd.) interval in the cutover area in each ordinal direction. Eight herb and four shrub plots will

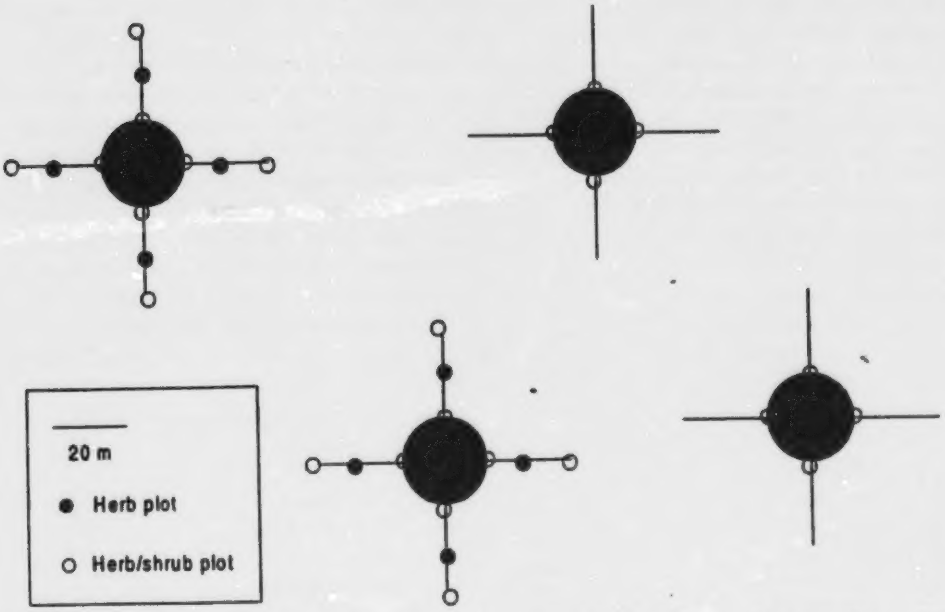

20 m
● Herb plot
○ Herb/shrub plot

FIGURE 5.1. Herb and shrub sampling design for small clumps. Survey plots will be located at 10 m (11 yd.) intervals along ordinal axes within each sampled clump.

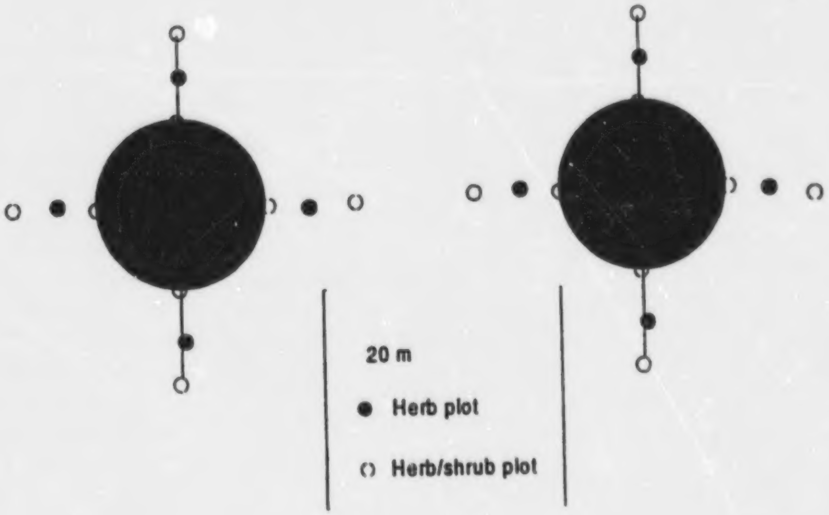

20 m
● Herb plot
○ Herb/shrub plot

FIGURE 5.2. Herb and shrub sampling design for medium clumps. Survey plots will be located at 10 m (11 yd.) intervals along ordinal axes from each sampled clump.

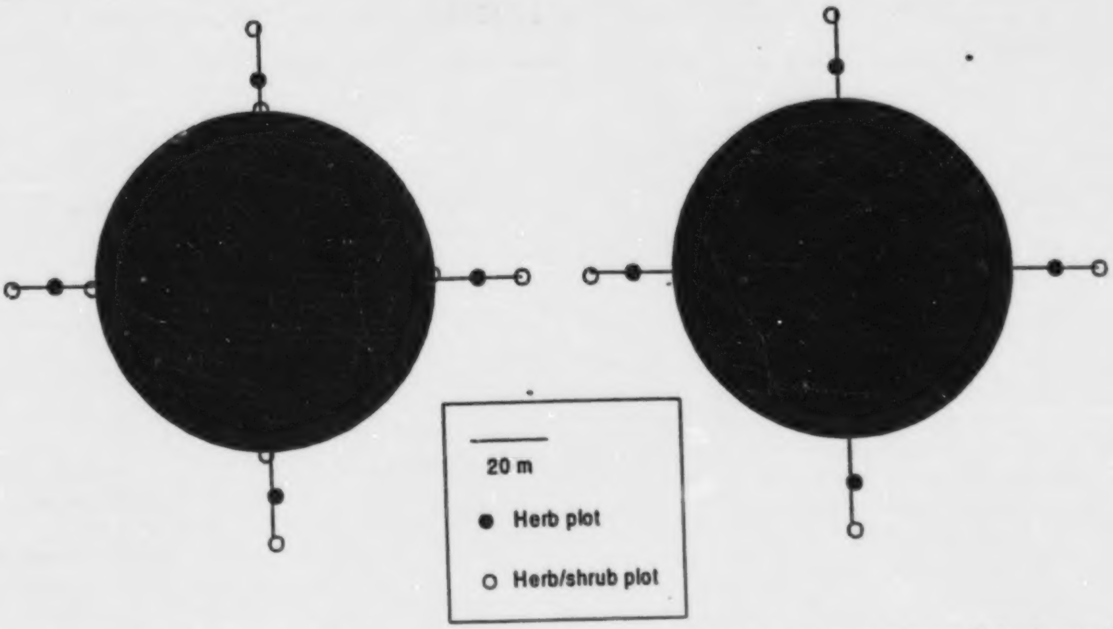


FIGURE 5. 3. Herb and shrub sampling design for medium clumps. Survey plots will be located at 10 m (11 yd.) intervals along ordinal axes from each sampled clump.

be sampled at a second large clump in each stand, but only at plots radiating into the cutover area.

In addition to the herb and shrub survey within and around clumps, each stand will have 20 herb plots and 10 shrub plots located randomly, but >50 m (>55 yd.), from any clump edge. This sampling scheme results in a total of 56 herb and 38 shrub plots sampled in each stand with small clumps, 54 herb and 28 shrub plots sampled in each stand with medium clumps, and 53 herb and 27 shrub plots sampled in each stand with large clumps.

Literature cited

Moss, E.H. 1992. Flora of Alberta, 2nd edition. Revised by J.G. Packer. University of Toronto Press, Toronto.

Vitt, D.H., J.E. Marsh, and R.B. Bovey. 1988. Mosses, lichens & ferns of northwest North America. Lone Pine Publishing, Edmonton, Alberta.

CHAPTER 6. ABUNDANCE AND SPECIES COMPOSITION OF CARABID BEETLES IN RELATION TO THE DISPERSION OF RESIDUAL TREES FOLLOWING HARVEST

Troy C. Sorensen

Introduction

Forest managers are beginning to recognize the importance of residual material in maintaining biodiversity within harvested areas. The question remains of how to spatially distribute this material within cutover areas to retain native biodiversity. Some spatial configurations may help to initially retain biodiversity within harvested stands (resistance), while other configurations may help harvested areas to quickly regain pre-disturbance biodiversity (resilience). One major, although often overlooked, component of biodiversity is beetles (Coleoptera).

One of the largest families of beetles in the world is carabidae (ground beetles). Approximately 946 species of carabids have been identified in Canada, of which 386 species are found in Alberta (Bousquet 1991). Almost all areas of Alberta are inhabited by carabids due to the large number of species and diverse habitat preferences. Adults range from 1.2 to 35 mm (0.05 – 1.4 in.) length (White 1983) and are usually carnivorous or omnivorous, feeding on live prey, carrion, and plant material. Carabid beetles can have two types of wing morphology; brachypterous (short-winged) species are assumed to be unable to fly; macropterous (long-winged) species are assumed to fly during migration and dispersal; and dimorphic species have both of the above morphs within their populations.

Recent studies have suggested that ground beetles may be sensitive to forestry practices and serve as ecological indicators (Thiele 1977; Holliday 1991, 1992; Halme and Niemelä 1993; Butterfield et al. 1995; Spence et al. 1996). Carabid communities are sensitive to temperature (Ericson 1979), moisture (Niemelä et al. 1992; Niemelä and Spence 1994), ground litter (Greenslade 1964; Guillemain et al. 1997), understory and canopy cover (Niemelä and Spence 1994), and are dependent on the availability of many other invertebrates as prey (Guillemain et al. 1997). Furthermore, the dispersion of suitable habitat can influence community composition because dispersal is energetically expensive (Atkins 1969) and species differ in their ability to fly (Thiele 1977).

The main objective of this project is to determine the influence of clump size and inter-clump spacing on the abundance and species composition of carabid beetles in aspen-dominated and pine-spruce-dominated forests of western Alberta. Specifically, I will examine 1) whether the carabid community within and surrounding residual clumps varies among stand treatments (see Chapter 3), and (2) whether differences among treatments can be explained by habitat preferences or dispersal capabilities (flight) of specific species.

Methods

Carabid beetles were surveyed with pitfall traps. Pitfall traps sample active beetles rather than all beetles and therefore catches over short durations are dependent on weather and the period of life cycle (Greenslade 1964; Ericson 1979). However, if traps are open for most of the snow-free portion of a year, the effect of weather should be negligible and catches should be correlated with beetle density (Baars 1979; Ericson 1979).

Each pitfall trap consisted of a removable 500 ml (0.53 qt.) plastic container set inside a larger [1 L (1.06 qt.)] permanent container (Spence and Niemelä 1994; Digweed et al. 1995). This trap design facilitated repeated sampling without disturbing the soil and litter around the trap. Each trap was filled 2-3 cm (0.8-1.2 in.) with undiluted ethylene glycol to increase trap effectiveness and preserve the specimens

(Holopainen 1992). A 15 x 15 cm (5.9 x 5.9 in.) plywood cover was propped over each trap with nails to prevent rain, leaves, and litter from falling into the trap.

Four replicate stands for each of the 3 harvesting treatments (see Chapter 3) were sampled in each study area in 1998. Four adjacent stands of unharvested forest were also surveyed within each study area as benchmarks for data from harvested stands. Traps were placed along radial transects from the center of residual clumps in harvested stands, and along similar transects in the unharvested stands (Fig. 6.1). Transects began at the center of the clump and ended at the halfway point between clumps in the cutover areas. The number of traps in each stand were similar (16-18); however the number of traps used to sample each clump, and the number of clumps sampled within each stand, differed among treatment (Fig. 6.1). Traps were placed > 20 m (21.9 yd.) apart to insure independent catches (see Digweed et al. 1995) which meant that some traps had to be placed alone to sample smaller clumps (Fig. 6.1). Traps were installed two weeks prior to sampling so that the initial litter and soil disturbance would not influence catches (Digweed et al. 1995). Traps were checked every two weeks from mid May (ground thaw) to the middle of August for a total of six trapping sessions. Catches from each trap were stored separately in vials with 75% ethyl alcohol. Species were identified according to Lindroth (1961-1969) and all were cross-referenced with collections at the Strickland Museum, University of Alberta, and with taxonomists at the University of Alberta.

Data Analyses

Beetle abundance was calculated as the number of beetles caught in undisturbed traps divided by the number of undisturbed traps. To estimate the abundance of beetles at the stand level, the abundances from each trap were weighted proportionally to the area of the stand that the trap represented (i.e. traps within cutover areas were given a weight of 0.97 and traps within

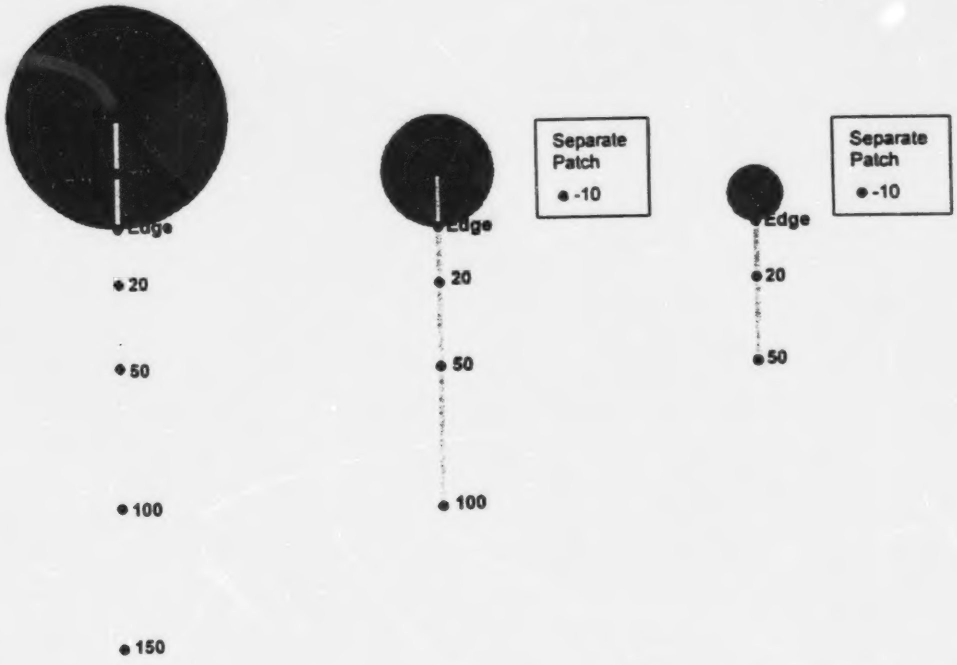


FIGURE 6.1. Locations of pitfall traps for the three clump sizes in the Whitecourt Area. Trapping design was similar for the Grande Prairie study area with the exception of a 50 m (55 yd.) trap in the center of large clumps.

clumps given a weight of 0.03. The proportion of traps disturbed within each two week shift increased over the summer (Fig. 6.2 a and b). Therefore only shifts 1-4 (<15% of traps disturbed) from each area were used in abundance and community analyses. The abundances of common species (i.e. species caught more than 50 times within each area; Halme and Niemelä 1993) were compared using ANOVA's among the four treatments. Post hoc comparisons were done with Tukey HSD (SPSS 1997). Standard errors of the mean are presented for all abundances and proportions.

Species richness within each stand was standardized with rarefaction methods (Krebs 1989). Standardized species richness were then compared among stand treatments using ANOVA's.

Detrended canonical correspondence analyses (DCCA) were used to examine community relationships among the treatments (Jongman et al. 1995). Data was standardized so that all species were given equal weight. This was necessary because species abundances cannot be compared due to differences in catchability (Halsall and Wratten 1988). Stand scores on the first two ordination axes were statistically compared with Monte Carlo permutation tests (1000 iterations, ter Braak 1987).

Canopy preferences (closed/open) and wing morphology (brachypterous/macropterous) for each species was obtained from Lindroth (1961-1969). The proportion of closed-canopy beetles and the proportion of brachypterous beetles among the four treatments were compared using ANOVA's.

Results

A total of 5386 carabid beetles of 31 species were caught in the Grande Prairie study area (Table 6.1). None of the species were exclusive to forested stands, however 13 species were absent from forested stands (Table 6.1). In shifts 1-4, 4601 carabids of 31 species were caught. Of these species, 10 were considered common.

A total of 4673 carabid beetles of 24 species were caught in the Whitecourt study area (Table 6.1). None of the species were exclusive to forested areas; however 14 species were found only in harvested stands (Table 6.1). In shifts 1-4, 3990 carabids of 24 species were caught. Of these species, 9 were considered common.

Abundance

Overall abundance of beetles was 2.23 ± 0.12 beetles/trap in the Grande Prairie area and 1.92 ± 0.15 beetles/trap in the Whitecourt area. In both study areas, the abundance of beetles was higher within clumps than within surrounding cutover areas or adjacent forest (Fig. 6.3 a and b). There was no significant difference in the weighted abundance of beetles among stand treatments in either study area (Grande Prairie, $F_{3,9} = 0.3$, $p = 0.83$; Whitecourt, $F_{3,9} = 0.8$, $p = 0.54$); however, abundance was highest within stands having small clumps in both areas (Fig. 6.4). This appeared to be due to a more even spatial distribution of beetles in stands with small clumps (Fig. 6.3 a and b).

Of the 10 common species in the Grande Prairie area, *Scaphinotus marginatus* ($F_{7,34} = 3.5$, $p = 0.01$) and *Pterostichus adstrictus* ($F_{7,34} = 2.8$, $p = 0.03$) had significant habitat associations; while *Platynus decentis* ($F_{7,34} = 2.1$, $p = 0.08$) and *Pterostichus punctatissimus* ($F_{7,34} = 2.1$, $p = 0.09$) had near significant associations (Fig. 6.5). The abundance of *S. marginatus* was higher within medium sized clumps than within all harvested areas (Fig. 6.5). *Loricera pilicornis* had higher abundance within intact forest, while *Agonum retractum*, *Pterostichus adstrictus*, *P. punctatissimus*, and *A. gratiosum* had highest abundances in harvested areas. Four species had highest abundances within residual clumps: *Platynus decentis*, *Scaphinotus marginatus*, *Carabus chamissonis*, and *Calathus ingratus* (Fig. 6.5).

Of the 9 common species in the Whitecourt area, *Trechus oregonensis* ($F_{7,21} = 2.8$, $p = 0.03$), *Scaphinotus marginatus* ($F_{7,21} = 2.5$, $p = 0.05$), *Pterostichus brevicornis* ($F_{7,21} = 4.4$, $p < 0.01$), and *P. adstrictus* ($F_{7,21} = 5.2$, $p < 0.01$) had significant habitat associations; although *Calathus advena* ($p = 0.06$) and *Pterostichus haematopus* ($p = 0.06$) had near significant associations (Fig. 6.6). *Trechus oregonensis* had highest abundance in the intact forest, while *S.*

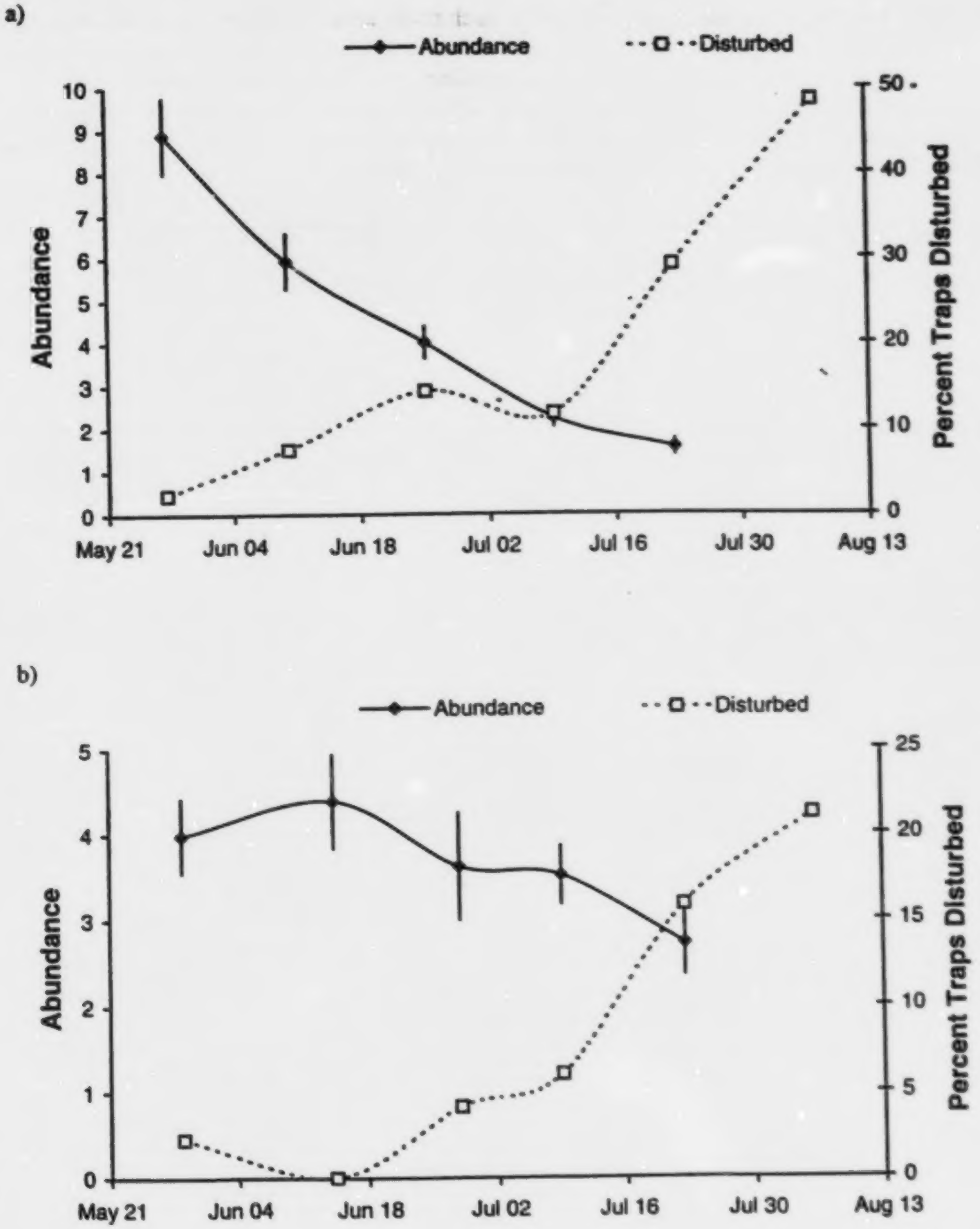


FIGURE 6.2. Abundances of beetles and disturbance of traps during 1998 in the Grande Prairie (a) and Whitecourt (b) study areas. The median date for each 2 week survey is plotted.

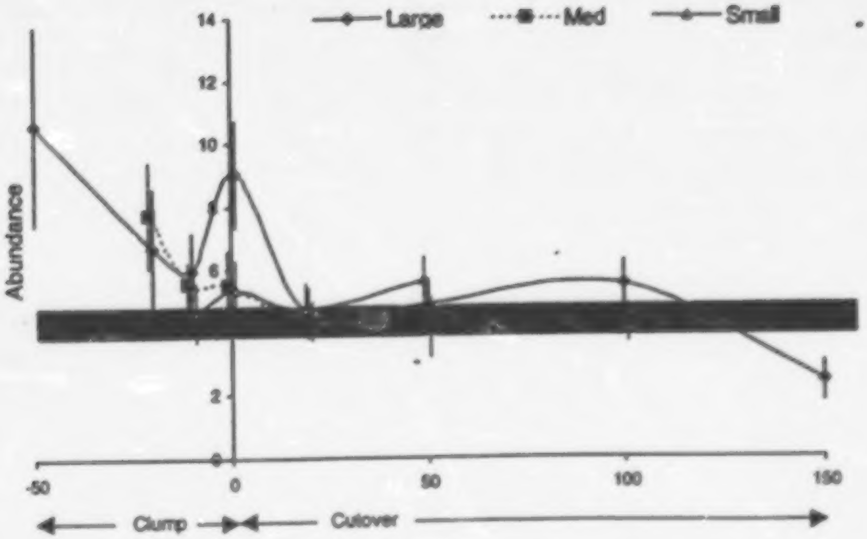
TABLE 6.1. Number of carabid beetles caught in each study area. Numbers caught are based on 15,994 trap-nights in the Grande Prairie area and 17,386 trap-nights in the Whitecourt area (May 21 - August 2, 1998). Flight capability (wing morphology) and canopy preferences were obtained from Lindroth (1961-1969). Some habitat descriptions were ambiguous or suggested that the species has no strong canopy association (left blank in table). Whether or not the species was found within adjacent unharvested stands is also indicated.

Study Area	Species	Number Caught	Flightless	Canopy	Found in Forest
Grande Prairie	*Pterostichus adstrictus*	1297	No	open	Yes
	Platynus decentis	1247	Yes	closed	Yes
	Pterostichus pensylvanicus	974	No	closed	Yes
	Scaphinotus marginatus	656	Yes	closed	Yes
	Agonum retractum	295	No	closed	Yes
	Calathus ingratus	276	Yes	closed	Yes
	Agonum gratiosum	101	Yes	open	Yes
	Pterostichus punctatissimus	90	Yes	closed	Yes
	Carabus chamissonis	60	Yes	closed	Yes
	Loricera pilicomis	60	No	closed	Yes
	Patrobus foveocollis	54	Yes	closed	Yes
	Calathus advena	42	Yes	closed	Yes
	Calasoma frigidum	35	No	closed	Yes
	Agonum cupreum	31	No	open	Yes
	Agonum sordens	26	No	closed	Yes
	Synuchus impunctatus	23	No	closed	Yes
	Platynus mannerheimi	22	Yes	closed	Yes
	Carabus meander	22	Yes	open	Yes
	Pterostichus haematopus	14	Yes	closed	No
	Agonum placidum	13	No	open	No
	Elaphrus clairvillei	10	No	closed	No
	Trechus oregonensis	10	Yes	closed	Yes
	Agonum affine	6	No	closed	No
	Pterostichus riparius	5	Yes	closed	No
	Harpalus fuivilabrus	5	Yes		Yes
	Agonum cupripenne	5	No	open	No
	Patrobus stygicus	3	No	closed	No
	Patrobus septentrionis	1	No	open	No
	Pterostichus melanarius	1	No	closed	No
	Badister neopulchellus	1	No	closed	No
	Chlaenius altematus	1	No		No

TABLE 6.1 cont.

Study Area	Species	Number Caught	Flightless	Canopy	Found In Forest
Whitecourt	*Scaphinotus marginatus*	1577	Yes	closed	Yes
	Calathus advena	1420	Yes	closed	Yes
	Pterostichus adstrictus	378	No	open	Yes
	Pterostichus riparius	368	Yes	closed	Yes
	Calathus ingratus	293	Yes	closed	Yes
	Platynus decentis	181	Yes	closed	Yes
	Pterostichus haematopus	190	Yes	closed	Yes
	Trechus oregonensis	99	Yes	closed	Yes
	Pterostichus brevicornis	61	Yes		Yes
	Patrobus foveocollis	52	Yes	closed	Yes
	Platynus mannerheimi	13	Yes	closed	No
	Agonum retractum	11	No	closed	Yes
	Nebria intermedia	10	Yes		Yes
	Pterostichus punctatissimus	7	Yes	closed	No
	Loricera pilicornis	6	No	closed	No
	Agonum cupreum	3	No	open	No
	Elaphrus clairvillei	3	No	closed	No
	Poecilus lucublandus	2	Yes	open	No
	Agonum placidum	2	No	open	No
	Miscodera arctica	1	No	open	No
	Calasoma frigidum	1	No	closed	No
	Calasoma calidum	1	No	open	No
	Synuchus impunctatus	1	No	closed	No
	Agonum graticsum	1	Yes	open	No

a)



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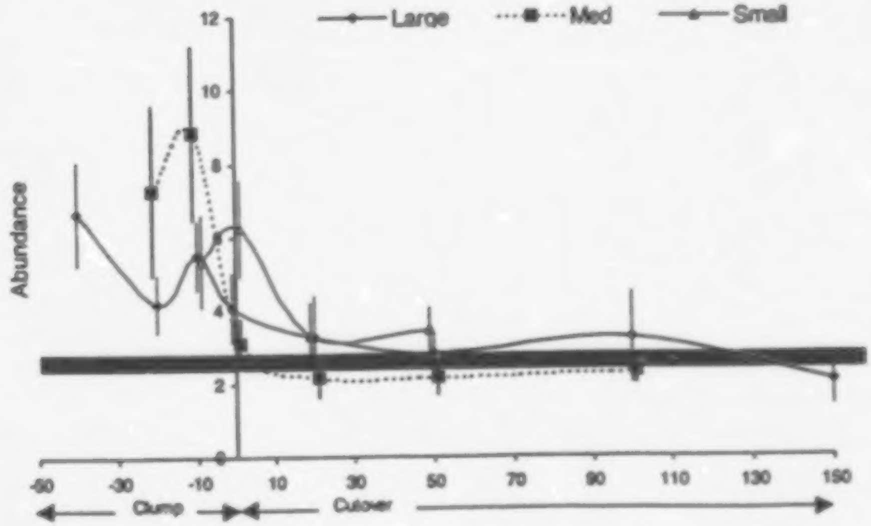


FIGURE 6.3 Abundance of beetles along transects in the Grande Prairie (a) and Whitecourt (b) study areas. The average abundance (± SE) of beetles within adjacent forest is represented by gray horizontal bars.

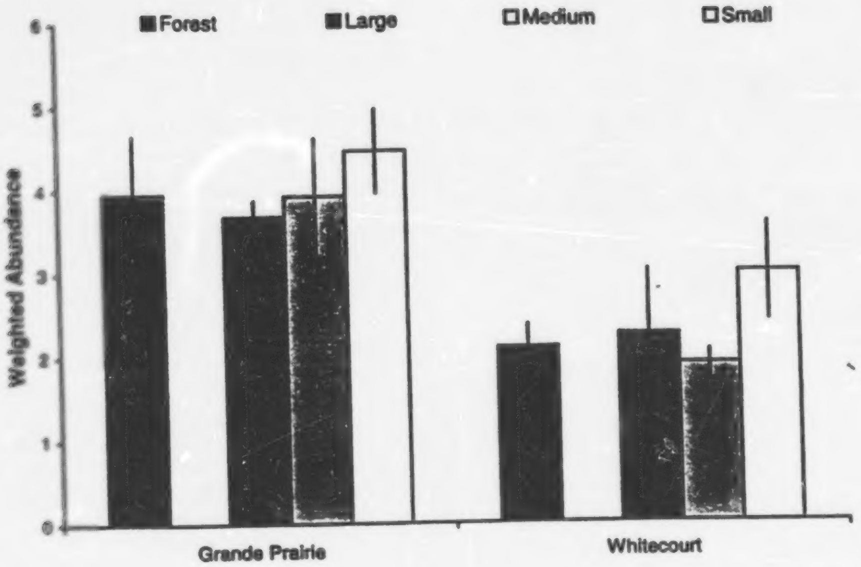


FIGURE 6.4. Abundance of beetles in the three stand treatments and adjacent old forest in both study areas. Abundances for each trap were weighted proportionally to the area of stand they represent.

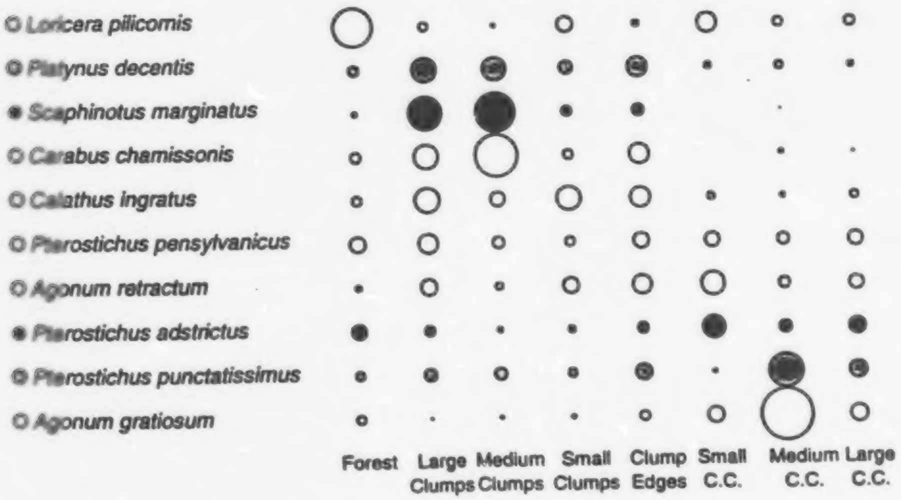


FIGURE 6.5. Relative abundances of the 10 common species of beetles in the Grande Prairie area. The proportion caught of each species is represented by the width of the circles. Dark circles represent species that differ among habitats ($p \leq 0.05$). Shaded circles represent species that may differ among habitats ($p \leq 0.10$). "C.C." refers to cutover areas (e.g. "Large C.C." refers to cutover areas in stands with large clumps).

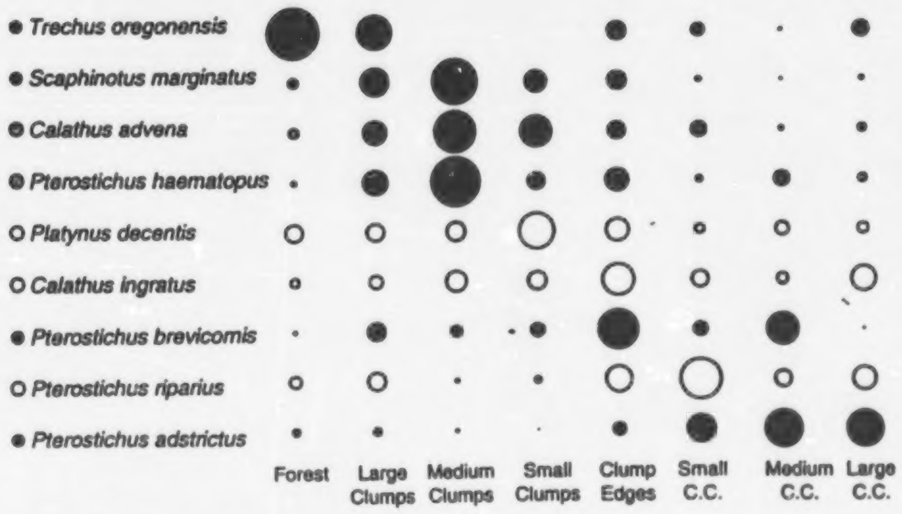


FIGURE 6.6. Relative abundances of the 9 common species of beetles in the Whitecourt area. The proportion caught of each species is represented by the width of the circles. Dark circles represent species that differ among habitats ($p \leq 0.05$). Shaded circles represent species with near significant habitat preferences ($p \leq 0.06$). "C.C." refers to cutover areas (e.g. "Large C.C." refers to cutover areas in stands with large clumps).

marninatus, *C. advena*, and *P. haematopus* had highest abundance in medium sized clumps (Fig. 6.6). *Platynus decentis* had highest abundances within small clumps (Fig. 6.6) but was also abundant in other clump sizes and in the adjacent forest. Three species had high abundance at clump edges and in cutover areas: *C. ingratus*, *P. brevicornis*, *P. riparius*, and *P. adstrictus* (Fig. 6.6).

Species Richness

In both study areas the standardized number of species was highest within stands with small clumps, and lowest within forested areas (Grande Prairie, $F_{3,11} = 3.7$, $p = 0.04$; Whitecourt, $F_{3,12} = 0.7$, $p = 0.57$, Fig. 6.7). Many species were found only in harvested stands and none were exclusive to adjacent forest (Table 6.1).

Community

In the Grande Prairie area, stands with large clumps had very similar community compositions to adjacent forested stands while medium sized clumps were least similar (Fig. 6.8a); however, stand treatments (including adjacent forest) did not explain significant variation. Stands with medium sized clumps were also least similar to adjacent forest in the Whitecourt area ($p = 0.05$). Stands with large and small clumps in the Whitecourt area were very similar to adjacent forests (Fig. 6.8b).

When only the data from *within* clumps was analyzed, the community of beetles within small clumps was least similar to communities in adjacent forests in both study areas (Fig. 6.9 a and b). In the Grande Prairie area, communities in medium and large sized clumps were most

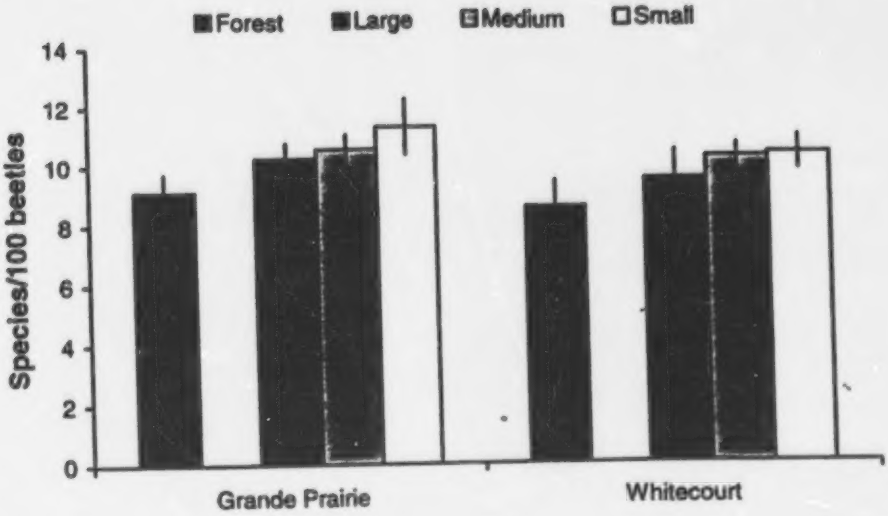


FIGURE 6.7. Standardized species richness in the three stand treatments and the adjacent old forest.

similar to communities in forested stands ($p = 0.38$, Fig. 6.9a). In the Whitecourt area, communities in large clumps were the most similar to communities in forested stands ($p < 0.005$, Fig. 6.9b).

The beetle community *outside* of clumps in the surrounding cutover areas shows a different pattern among stand treatments. In the Grande Prairie area, cutover areas around medium sized clumps were most similar to adjacent forest, cut-over areas around small clumps were least similar ($p = 0.15$, Fig. 6.8a). In the Whitecourt area, cutover areas around large or small clumps were most similar to adjacent forest, while cut-over areas around medium sized clumps were least similar ($p = 0.05$, Fig. 6.10b).

In both areas, the proportion of beetles classified as preferring closed-canopies was highest within clumps, and lowest in cutover areas (Fig. 6.11 a and b). In the Grande Prairie area, the proportion of beetles classified as preferring closed-canopies was highest in stands with large clumps. In the Whitecourt area, the proportion was highest in stands with small clumps (Fig. 6.12), likely due to a more even spatial distribution of these species in the Whitecourt study area (Fig. 6.11b).

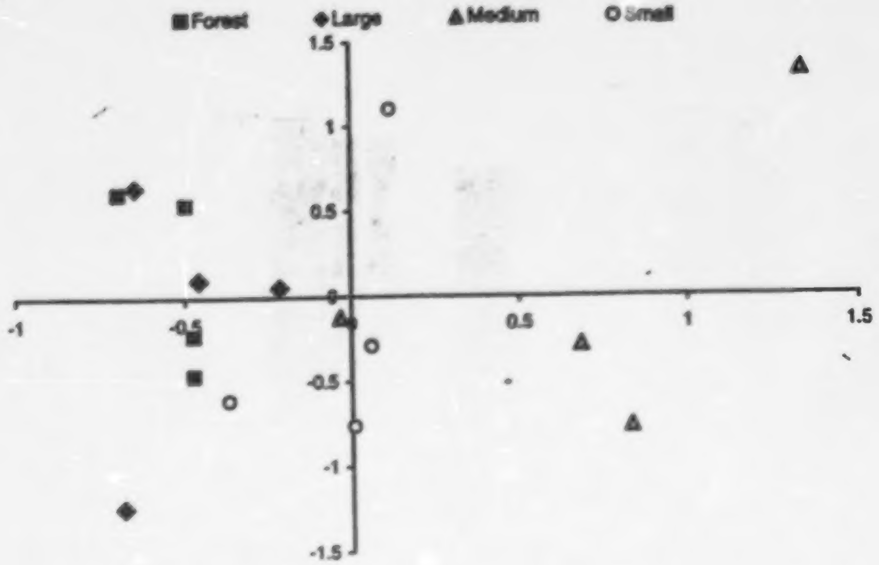
The proportion of beetles unable to fly (brachypterous) was generally higher within clumps than within adjacent forest or cutover areas in both study areas (Fig. 6.13 a and b). Flightless beetles occurred in higher proportions in cut-over areas with medium sized clumps in the Grande Prairie area (Fig. 6.13a). This resulted in a stand level effect where the proportion of flightless beetles was highest in stands with medium sized clumps (Fig. 6.14). In the Whitecourt area, cut-over areas with small clumps had the highest proportions of flightless beetles with an even spatial distribution (Fig. 6.13b). This resulted in stands with small clumps to have the highest proportion of flightless beetles (Fig. 6.14).

Discussion

Influence of Harvest

The abundance of beetles was higher within harvested stands than within adjacent forest. This can be explained by higher plant productivity and the abundance of herbivorous insects within open habitats and forest edges (Hahta 1976; see also Spence et al. 1996).

a)



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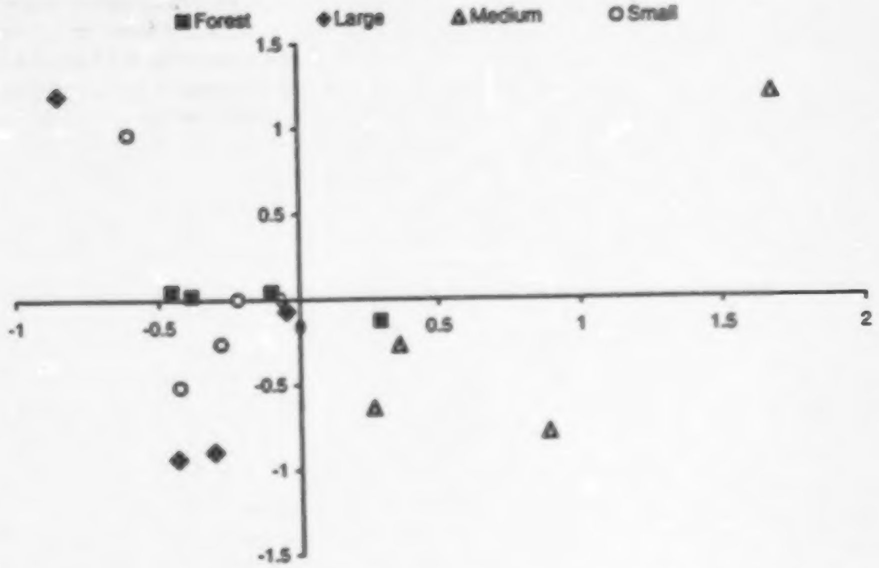
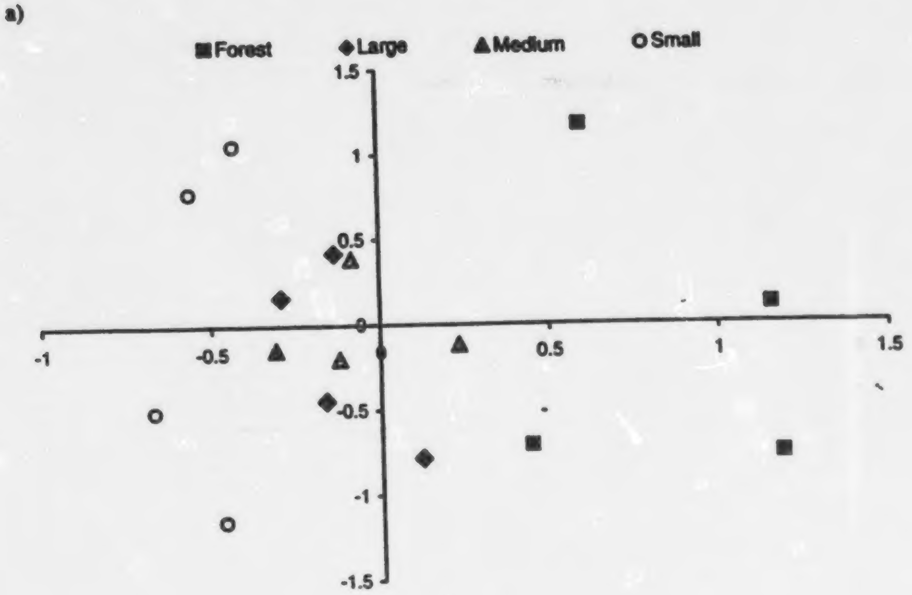


FIGURE 6.8. Detrended Canonical Correspondence Analysis (DCCA) of stands in the Grande Prairie (a) and Whitecourt (b) study areas.



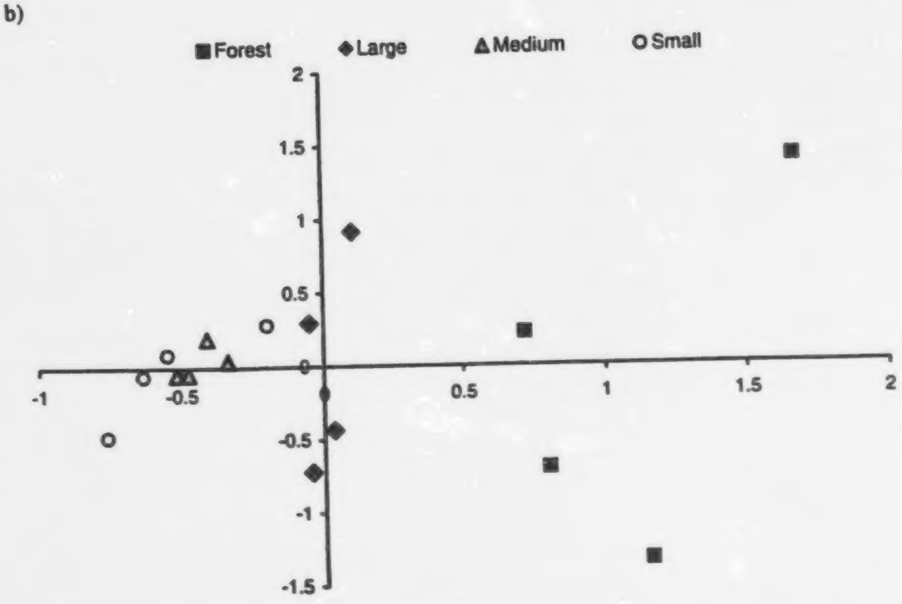


FIGURE 6.9. Detrended Canonical Correspondence Analysis (DCCA) of traps *within* clumps in the Grande Prairie (a) and Whitecourt (b) study areas.

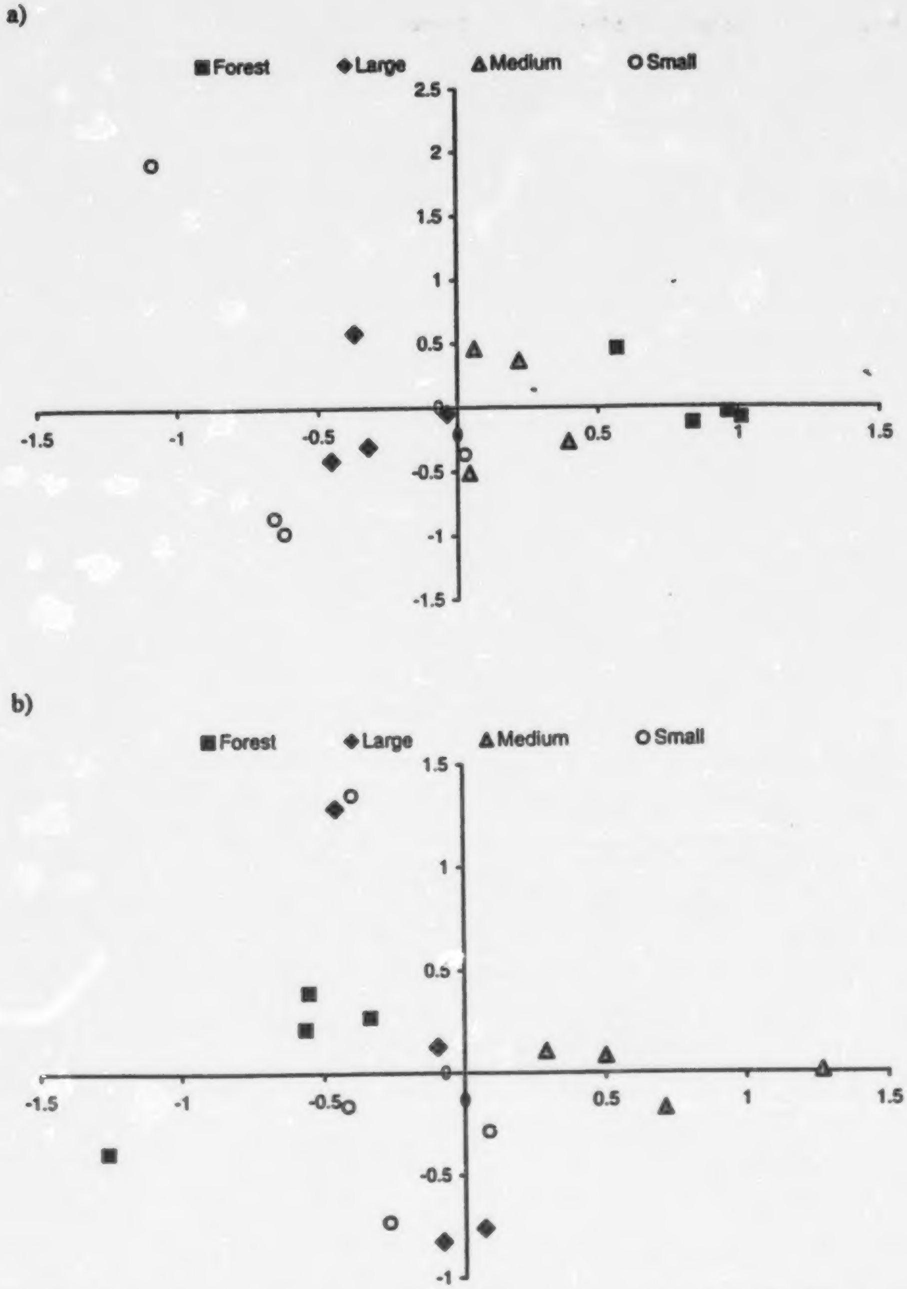
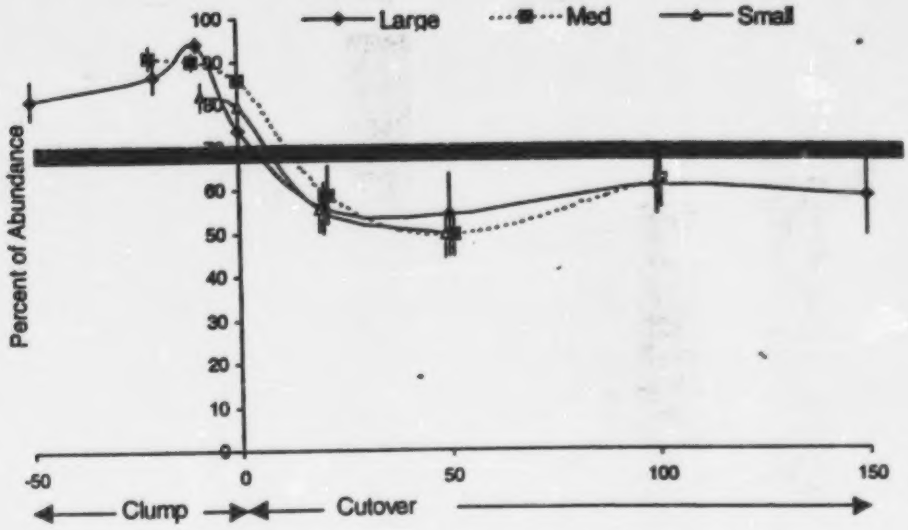


FIGURE 6.10. Detrended Canonical Correspondence Analysis (DCCA) of traps *outside* clumps in the Grande Prairie (a) and Whitecourt area (b) study areas.

a)



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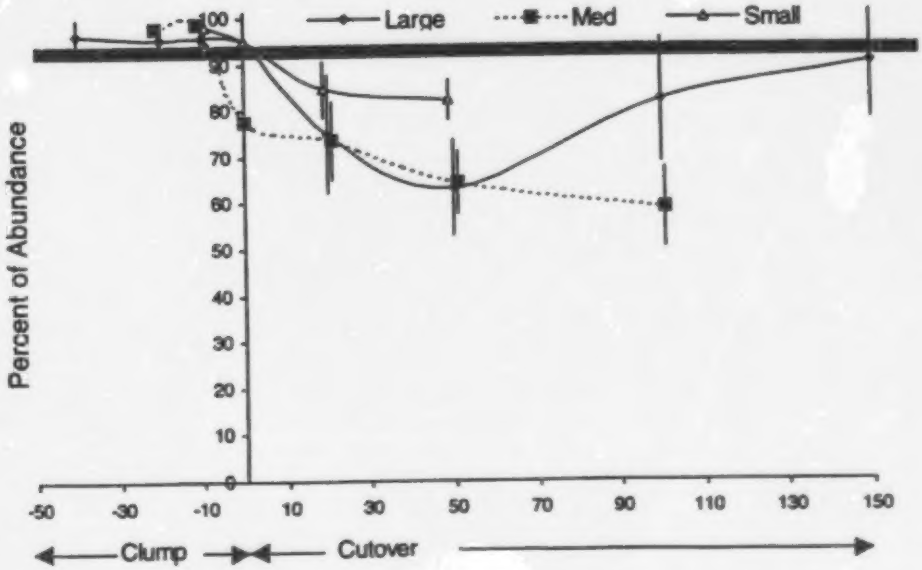


FIGURE 6.11. Proportion of beetles that were closed-canopy species along transects in the Grande Prairie (a) and Whitecourt (b) study areas.

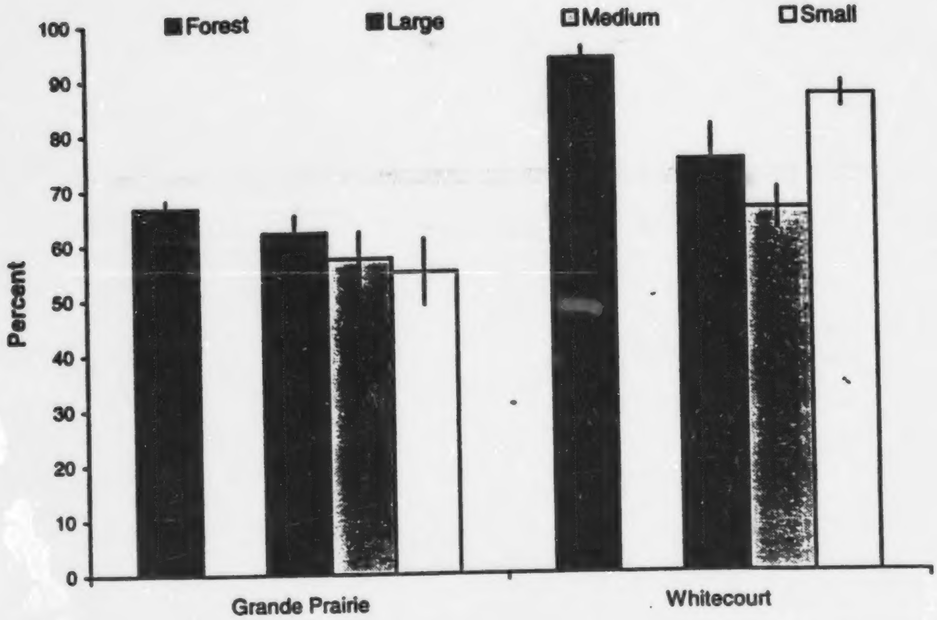


FIGURE 6.12. Proportion of beetles that were closed-canopy species along transects in the Grande Prairie and Whitecourt areas. Abundances for each trap were weighted proportional to the area of stand they represent.

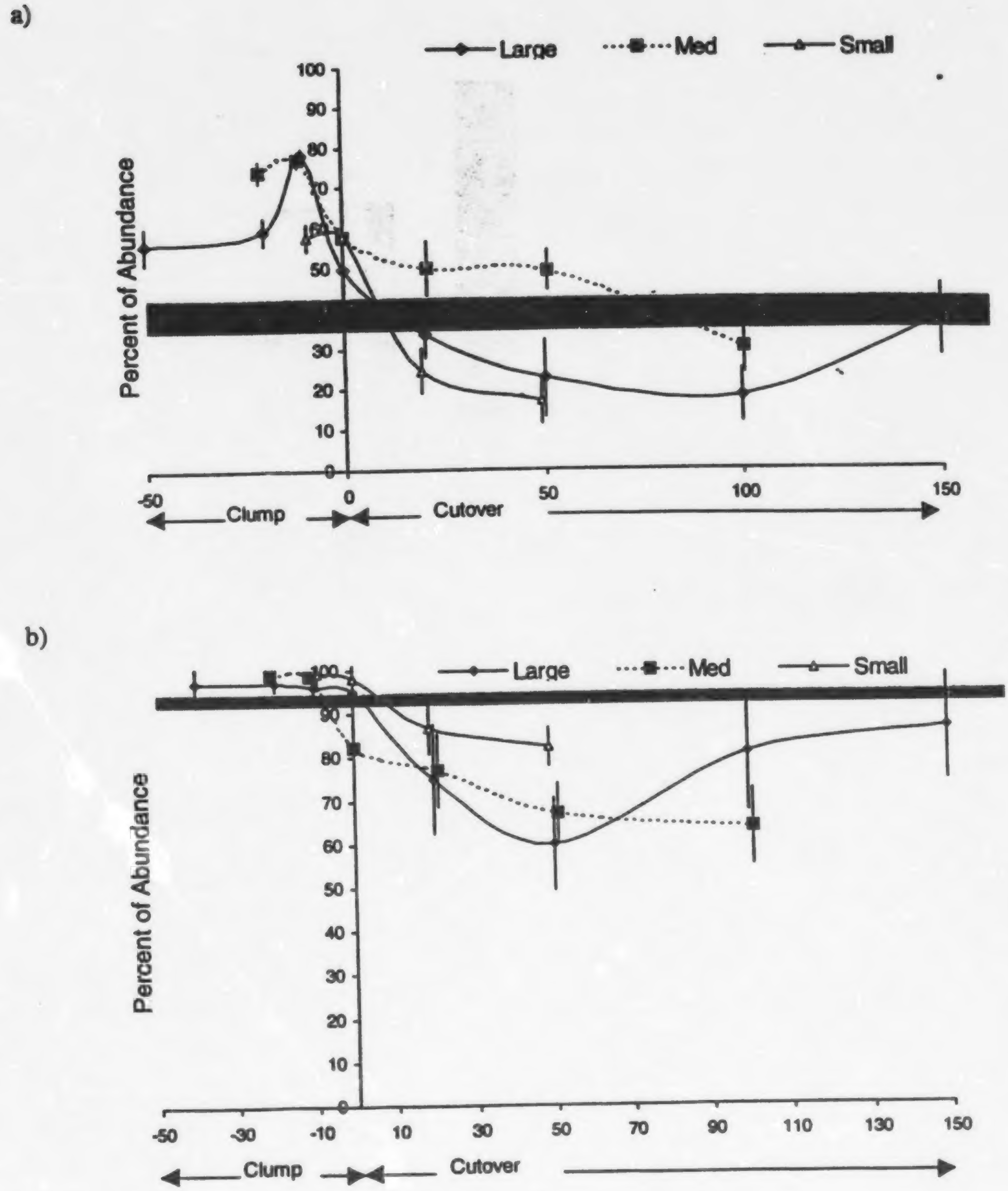


FIGURE 6.13. Proportion of beetles that were brachypterous species along transects in the Grande Prairie (a) and Whitecourt (b) study areas.

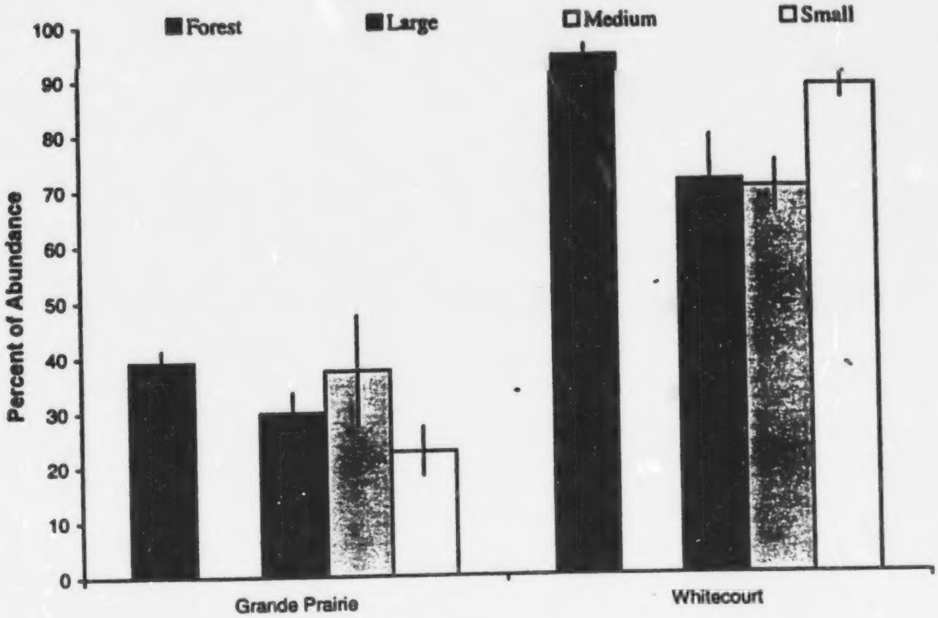


FIGURE 6.14. Proportion of beetles that were brachypterous species along transects in the Grande Prairie and Whitecourt areas.

Standardized species richness was also higher in harvested stands than adjacent forest (see also Spence et al. 1996). Open or disturbed habitats may have higher spatial heterogeneity than contiguous forest (see Halme and Niemelä 1993) which would result in a more diverse beetle community (i.e. higher pooled species richness or beta-diversity; see also Niemelä et al. 1988; Didham et al. 1998). This is also the case for most arthropods following disturbance (Jennings et al. 1988; Danks and Foottit 1989; McIver et al. 1992; Didham et al. 1998; Butterfield et al. 1995).

Influence of Harvest Residual

In both study areas, stands with many small clumps had the highest carabid abundance and species richness (see also Halme and Niemelä 1993). This is probably due to high spatial heterogeneity within the stand, and high productivity along clump edges. In addition, the abundance of carabids was higher within clumps than within adjacent forest. This could also be due to habitat heterogeneity and higher plant productivity along edges; however carabid abundance in large clumps was just as high in the middle of the clump as it was at the edge. Perhaps beetles shifted their home range to incorporate clumps following harvest, and were therefore crowded within clumps at abundances higher than would occur in an unfragmented matrix. Consequently, abundance may decrease during the next few years if these beetles are not able to survive while dispersing between clumps, or if the resources within clumps is insufficient to support breeding populations (Niemelä et al. 1988).

Halme and Niemelä (1993) suggested that open habitat species were more likely to invade smaller clumps than larger clumps. In both areas of the current study, large clumps retained communities most similar to adjacent forests while communities in small clumps were least similar to adjacent forests. Large clumps probably provide the microhabitat and prey that forest carabids prefer. On the other hand, smaller and more abundant clumps appear to draw forest communities out into the cutover areas of stands. Carabids may have difficulty traveling between large clumps in the current study with an inter-clump distance of 300 m (328 yd.). Averages for *Calathus* and *Pterostichus* suggest that single beetles only travel 66 m (72 yd.) to 160 m (175 yd.) a season, respectively (Baars 1979). Furthermore, carabid species that are restricted to contiguous, undisturbed forest are often large-bodied species with low dispersal abilities (Halme and Niemelä 1993). Although small clumps may not contain high abundances of forest species, stands with many small clumps appear to have clumps close enough together to allow forest species to travel through or forage in cutover areas.

Den Boer (1970) found that brachypterous species were less able to reach isolated habitats than dimorphic or macropterous species. Thus, the proportion of brachypterous species should be lower in clumps that are more isolated. However, in the current study most beetles were probably not directly killed by harvesting and may have moved into clumps upon awaking from hibernation. This would explain the high proportion of flightless beetles within patches.

Management Implications

Results from this study provide several management tools. Clumps of residual trees can be used to maintain forest communities of carabid beetles within harvested stands in the first growing season. At the same time, the abundance of carabid beetles should be very high within residual clumps; however it is currently unknown if these elevated abundances are sustainable over the long-term.

The dispersion of residual clumps within stands can also be manipulated for various objectives. Clump size can be maximized to aid in the retention of forest communities within clumps ("lifeboats"), and clump density can be maximized to make carabid communities in cutover areas more similar to adjacent forest. Larger clumps appear to contain habitat more similar to intact forest, while higher clump densities aid in the movement of forest species within stands.

Future Research

Although results from this study highlight how carabid beetles are initially influenced by harvesting and the dispersion of residual material, only longer-term studies will document whether beetle communities return to rotation or post-rotation states. Thus, carabid beetle communities should be resurveyed throughout succession. These subsequent surveys would be most informative at the following time periods: 6-10 yr, 10-15 yr, 30-40 yr, and 60-80 yr.

At 6-10 yr post-harvest trees become established and arthropods often increase in abundance and diversity (Huhta 1971, 1976). This increase may be due to high prey abundance resulting from high plant productivity. Furthermore, beetle population dynamics should have stabilized within clumps, and open habitat species will have had time to establish. By 10-15 years post-harvest the canopy will be closed and open habitat species may begin losing dominance in the community (Huhta 1971; McIver et al. 1992). Furthermore, beetle habitat may be influence by the complete decomposition of small DWM generated by harvest (see Chapter 4). By 30-40 years the forest floor will be shaded and hygrophilous species may begin to colonize the harvested areas. Furthermore, natural thinning of young trees will be adding DWM to the forest floor, which is important habitat for beetles. Beetle surveys at 60-80 years will determine whether or not carabid communities can recover within one rotation.

Literature Cited

Atkins, M.D. 1969. Lipid loss with flight in the Douglas-Fir beetle. The Canadian Entomologist **101**: 164-165.

Baars, M.A. 1979. Catches in pitfall traps in relation to mean densities of carabid beetles. Oecologia (Berl.) **41**: 25-46.

Bousquet, Y. 1991. Checklist of beetles of Canada and Alaska. Research Branch, Agriculture Canada, Publication 1861-E.

Butterfield, J., Luff, M.L., Baines, M., and Eyre, M.D. 1995. Carabid beetle communities as indicators of conservation potential in upland forests. Forest Ecology and Management **79**: 63-77.

Danks, H.V., and Foottit. R.G. 1989. Insects of the boreal zone of Canada. Canadian Entomologist **121(8)**: 625-690.

Digweed, S.C., Currie, C.R. Cárcamo, H.A. and. Spence, J.R 1995. Digging out the "digging-in effect" of pitfall traps: influences of depletion and disturbance on catches of ground beetles (Coleoptera: Carabidae). Pedobiologia **39**: 561-576.

Ericson, D. 1979. The interpretation of pitfall catches of *Pterostichus cupreus* and *Pt. melanarius* (Coleoptera, Carabidae) in cereal fields. Pedobiologia **19**: 320-328.

Greenslade, P. J. M. 1964. Pitfall trapping as a method for studying populations of Carabidae (Coleoptera). Journal of Animal Ecology **33**: 301-310.

Guillemain, M., Loreau, M. and Daufresne, T. 1997. Relationships between the regional distribution of carabid beetles (Coleoptera, Carabidae) and the abundance of their potential prey. Acta Oecologica **18**: 465-483.

Halme, E. and Niemelä, J. K. 1993. Carabid beetles in fragments of coniferous forest. Annales Zoologica Fennici **30**: 17-30.

Halsall, N.B., and Wratten, S.D. 1988. The efficiency of pitfall trapping for polyphagous predatory Carabidae. Ecological Entomology **13**: 293-299.

Holliday, N. J. 1991. Species responses of carabid beetles (Coleoptera; Carabidae) during post-fire regeneration of boreal forest. Canadian Entomologist **123**: 1369-1389.

Holliday, N. J. 1992. The carabid fauna (Coleoptera: Carabidae) during postfire regeneration of boreal forest: properties and dynamics of species assemblages. Canadian Journal of Zoology **70**: 440-452.

Holopainen, J. K. 1992. Catch and sex ratio of Carabidae (Coleoptera) in pitfall traps filled with ethylene glycol or water. Pedobiologia **36**: 257-261.

Huhta, V. 1971. Succession in the spider communities of the forest floor after clear-cutting and prescribed burning. Ann. Zool. Fennici **8**: 483-542.

Huhta, V. 1976. Effects of clear-cutting on numbers, biomass and community respiration of soil invertebrates. Ann. Zool. Fennici **13**: 63-80.

Jennings, D.T., Houseweart, M.W. Dondale, C.D. and Redner, J.H. 1988. Spiders (Araneae) associated with strip-clearcut and dense spruce-fir forest of Maine. Journal of Arachnology **16**: 55-70.

Lindroth, C. H. 1961-1969. The ground-beetles of Canada and Alaska: Parts 1-6. Opuscula Entomologica Supplementa **20**, **24**, **29**, **33**, **34**, **35**.

McIver, J.D., Parsons, G.L. and Moldenke, A.R. 1992. Litter spider succession after clear-cutting in a western coniferous forest. Canadian Journal of Forest Research **22**: 984-992.

Niemelä, J.K., Halme, E. Pajunen, T. and Haila, Y. 1986. Sampling spiders and carabid beetles with pitfall traps: the effect of increased sampling effort. Annales Entomologici Fennici **52**: 109-111.

Niemelä, J. K. and Spence, J. R. 1994. Distribution of forest dwelling carabids (Coleoptera): spatial scale and the concept of communities. Ecography **17**: 166-175.

Niemelä, J. K., Spence, J. R. and Spence, D. H. 1992. Habitat associations and seasonal activity of ground-beetles (Coleoptera, Carabidae) in central Alberta. The Canadian Entomologist **124**: 521-540.

Spence, J. R., Langor, D. W. Niemelä, J. K. Carcamo, H. A. and Currie, C. R. 1996.

Northern forestry and carabids: the case for concern about old-growth species. Annales Zoologica Fennici 33: 173-184.

Spence, J. R. and Niemelä, J. K. 1994. Sampling carabid assemblages with pitfall traps: the madness and the method. Canadian Entomologist 126: 881-894.

SPSS Inc. 1997. SPSS 7.5.1 for Windows.

Ter Braak, C.J.F. 1987. CANOCO: a FORTRAN program for canonical community ordination by correspondence analysis, principal components analysis and redundancy analysis (version 2.1). Agricultural Mathematics Group, Wageningen, 95pp.

Thiele, H.U. 1977. Carabid beetles in their environments. Springer-Verlag, Berlin.

White, R.E. 1983. A field guide to the beetles of North America. The Peterson Field Guide Series. Houghton Mifflin Company, New York.

CHAPTER 7. VARIATION IN BIRD COMMUNITIES IN RELATION TO THE NUMBER AND SIZE OF TREE CLUMPS RETAINED AT HARVEST

Jim Schieck

Introduction

As a result of government regulations (e.g. British Columbia Ministry of Forests 1995) and exploratory adaptive management (e.g., Alberta-Pacific Forest Industries Inc. 1996) many forest companies have modified harvesting strategies so that they leave standing trees and snags in harvested areas. By providing a more complex habitat than found after conventional clear-cut harvest, residual trees may make the structured cut-blocks (also called variable retention blocks; Franklin et al. 1997) more useable to native forest birds (Hansen et al. 1995a, Norton and Hannon 1997, Steventon et al. 1998). In addition, live trees that are retained at harvest may promote recovery of biota in harvested areas by providing inputs of large dead trees (snags) and downed logs throughout succession (Hansen et al. 1991, Schieck et. al. 1995, Lindenmayer and Franklin 1997).

Although not well studied, the pattern in which residual trees are retained also may affect bird communities in harvest areas. These patterns encompass a range, from single trees scattered throughout a cut-block, to all trees retained in a single clump. Due to external species (DellaSala and Rabe 1987, Saunders et al. 1991, Brothers and Spingarn 1992) and environmental conditions (Chen et al. 1995) penetrating the edge of clumps, large clumps probably retain more of the microhabitats and ecological process that were present in the forest pre-harvest than do small clumps (Lindenmayer and Franklin 1997, Weinberg and Roth 1998, but see Matthysen and Adriaensen 1998). Some forest birds require specialized forest structures and/or microhabitats for foraging and protection, and these birds may be present in large but not small residual clumps (DellaSala and Rabe 1987, Schmiegelow et al. 1997). However, if all trees and snags are retained in a single clump, the remainder of the cut-block may function like a traditional clear-cut. Retaining trees and snags throughout the cut-block may be more beneficial to birds that require only a few large trees for nesting or perching (Holt and Martin 1997). In addition, if scattered residual trees remain standing and grow throughout the next rotation they will provide forest structures that are associated with large trees in mid-rotation forests, and may result in more forest birds being able to live in mid-rotation age forests (Hansen et al. 1991, Schieck et al. 1995, Lindenmayer and Franklin 1997). Thus, the most appropriate distribution of trees and snags in structured cut-blocks may depend on the types of species that are present pre-harvest and the management objectives for the area.

Biotic and ecological differences that result from leaving residual trees in different patterns throughout cut-blocks has not been evaluated. Consequently, forest managers have been subjectively deciding which patterns to create (e.g., Alberta-Pacific Forest Industries Inc. 1996). Only by experimentally manipulating patterns of residual trees, and then studying birds, will we understand the impacts on birds of differing residual tree patterns. In this study, we established four replicates of three treatments (clump sizes and inter-clump distances differed among treatments) in two experimentally harvested areas. In this chapter I evaluate differences in bird communities among these treatments.

Objectives

During 1997 I evaluated whether bird communities varied among the treatments prior to harvest. Due to funding considerations, only cursory bird surveys were conducted during 1998. Five large, medium, and small residual tree clumps, and five cutover sites that were >100 m (>109 yd.) from residual trees were surveyed. In this chapter I compared bird communities found pre-harvest with those found during the first year post-harvest. Analyses were conducted separately for migrants versus residents, and for birds that do and do not nest or forage in large trees or snags, because birds from these categories were expected to be affected differently by the size and dispersion of residual tree clumps.

Intensive bird surveys will be conducted during 1999, both within the cut-blocks and within adjacent unharvested forest. I will conduct rigorous comparison of bird communities among clump sizes, cutover sites, and unharvested forest based on those data. In addition, based on a comparison of the bird communities pre-harvest, and those present in the first two years post-harvest I will evaluate year-to-year differences in bird communities. Results from all analyses will be included in the final report.

Methods

Study Design

A description of the study areas and experimental harvesting is presented in Chapter 2. Descriptions of the pre- and post-harvest trees, snags, and down woody materials, is presented in Chapters 3 - 4. A cursory description of the pre-harvest understory vegetation is presented in Chapter 5.

Depending on the shape of the stands, either two or three strip transects 100 m (109 yd.) wide and having a total length of 1 km (0.62 mi.) were established in each stand (Appendix II and IV). Transects were divided into ten 1-ha sites with permanent markers located at the center of each site. To avoid including birds that lived in adjacent habitats, the outer edge of the transects were a minimum of 60 m (65 yd.), and when possible > 100 m (> 109 yd.), from adjacent forest types. During 1998, 0.79 ha (2.0 acre), 50 m (55 yd.) radius circles were surveyed in residual clumps and cutover areas post-harvest. Cutover sites were > 100 m (109 yd.) from both the edge of the cut-blocks and residual tree clumps within the cut-blocks. Sites > 100 m (109 yd.) from residual trees could only be found in cut-blocks with large clumps and in cut-blocks that were not part of the study.

Bird Surveys

During 1997 birds were surveyed four times along transects in each stand at approximately bi-weekly intervals between 9 May and 26 June. During each survey, the center lines of the transects were walked at a rate of 100 m (109 yd.) per 10 min. Surveys were conducted between first light and three hours past sunrise on days with little wind and no rain. A 50 m (55 yd.) detection distance was used for these transects because that represents the maximum distance that all bird vocalizations could be detected reliably (Emlen and DeJong 1981, Wolf et al. 1995, Schieck 1997). At the center of each 1-ha (2.47-acre) site, a 5-minute point count was conducted.

Birds were expected to move into the forest clumps and adjacent unharvested forest immediately post-harvest (Lovejoy et al. 1986, Schmiegelow 1997). These relatively high densities of birds immediately post-harvest, however, are expected to decrease as some individuals move and others die (Lovejoy et al. 1986, Schmiegelow 1997). Thus, since we had financial constraints in this study, only cursory bird surveys were conducted during 1998 - only five residual tree clumps of each size, and five cutover sites were surveyed. Sites were surveyed within stand groups B and C in the Whitecourt study area, and within stand groups A and B in the Grande Prairie study area. Sites were surveyed 4 times at bi-weekly intervals between May 6 and June 15 using point counts (10 minute duration, 50 m (55 yd.) radius). The order that stands were surveyed during each period was randomly determined.

During bird surveys all birds heard or seen were plotted onto maps. Movements were summarized so that individual birds were tallied only once. Observers were trained, and methods were standardized prior to the start of the surveys. A few of the vocalizations we detected could not be identified to species during the surveys. These vocalizations were recorded using cassette tape recorders and parabolic microphones. Attempts were made to locate and identify unknown birds after the end of the surveys. The remaining unidentified vocalizations were identified by comparing them to recordings from known birds. Tree Swallows (see Tables 7.1 and 7.4 for scientific names) were included if they were detected flying within the point count or transect sites because they forage while flying (Godfrey 1986). Common Snipe that were conducting their aerial territorial display within the point count or transect sites were also included. Other species that were detected flying over the survey sites were not included.

Analyses

For each bird species, an index of abundance at each site was calculated as the mean number of individuals detected during the surveys. Indices of abundance were converted to indices of density by dividing by the area surveyed (i.e., dividing by 1.00 for the transect surveys, and by 0.79 for the point count surveys). Indices of species richness at each site were calculated as the mean number of species detected during a visit, divided by the area surveyed.

ANOVA was used to evaluate whether total bird density and species richness differed among treatment types pre-harvest. I did not evaluate whether indices of density for each species differed among clump sizes post-harvest because sample size was small. Bird species were categorized based on their migration habits (year-round residents versus migrants) and their use of large trees and snags for nesting and foraging (Campbell et al. 1990, Semenchuk 1992, Gauthier and Auby 1996, Campbell et al. 1997) and analyses were conducted separately for each of these categories.

Differences in bird species richness and density may not accurately portray differences among bird communities because species composition may differ even when richness and density are similar. Thus, I used correspondence analyses and canonical correspondence analyses (CA and CCA, respectively) to evaluate whether bird communities differed among treatment types pre-harvest and differed among residual clump sizes post-harvest. In all CAs and CCAs, the option to down-weight rare species was used (ter Braak 1992) because chance events may have resulted in rare species being observed in non-typical habitats. In all analyses, results were considered statistically significant if the probability of them occurring by chance was less than 0.05.

Results

Whitecourt Study Area

Pre-harvest - Within the 12 stands (120 sites) that were surveyed pre-harvest, 42 bird species and 901 individuals were detected (Table 7.1). This bird community was dominated by species that migrate with only 10 species (24%) being year-round residents. Most residents (70% of 10 species), but fewer migrants (40% of 32 species) nested or foraged in large live trees or snags, although that difference was not statistically significant ($\chi^2 = 2.6$, $P = 0.11$, $df = 1$).

Since vegetation in and around the stands differed among stand groups pre-harvest (Chapters 2-5), I evaluated whether it was necessary to control for differences among stand groups before determining whether bird species richness and density differed among treatment types. Bird density and richness were higher in stand group B than in the other three stand groups (Table 7.2). That difference was largely due to resident bird species that use large trees being at higher density in stand group B than in the other stand groups. Thus, I adjusted for differences among stand groups before evaluating whether differences occurred among treatments (i.e., the factor "stand group" was included in the analyses before the factor "treatment" and I used Type III sum of squares to evaluate differences among treatments, SAS Institute Inc. 1989). Within this analysis, bird density and species richness were similar among treatments for each of the categories of birds, and when all categories were combined (Table 7.2).

Similar results were found within the community analyses. The first and second functions of the CA had eigen values of 0.07 and 0.04, respectively. Combined, these two functions accounted for only 3.4% of the variation in bird species density. However, even with this small amount of explained variation, bird communities differed among stand groups pre-harvest ($F = 1.8$, $P < 0.001$; dummy variables coding stand groups were included as environmental variables in the CCA, and differences among stand groups were tested using a bootstrap Monte Carlo test with 999

TABLE 7.1. Species names, codes, migration habits, use of large trees for nesting or foraging, and density (mean ± SE) within pre- and post-harvest sites in the Whitecourt study area, that were surveyed during 1997 and 1998.

Common Name	Genus And Species	Species Code	Migration Habits	Use Large Trees	Density (number/10 ha)				
					Pre-Harvest Sites	Large Clumps	Medium Clumps	Small Clumps	Cutover Sites
Northern Goshawk	*Accipiter gentilis*	NOGO	Resident	Yes	0.04 ± 0.03	0	0	0	0
Ruffed Grouse	*Bonasa umbellus*	RUGR	Resident	No	0.02 ± 0.02	0	0	0	0
Kildeer	*Charadrius vociferus*	KILL	Migrant	No	0	0	1.91 ± 0.78	0	0
Common Snipe	*Gallinago gallinago*	COSN	Migrant	No	0.04 ± 0.03	3.17 ± 1.42	0.64 ± 0.63	2.54 ± 1.56	1.91 ± 0.78
Yellow-bellied Sapsucker	*Sphyrapicus varius*	YBSA	Migrant	Yes	0.17 ± 0.06	0	0	0	0
Downy Woodpecker	*Picoides pubescens*	DOWO	Resident	Yes	0.02 ± 0.02	0	0	0	0
Hairy Woodpecker	*Picoides villosus*	HAWO	Resident	Yes	0.08 ± 0.04	0	0	0	0
Three-toed Woodpecker	*Picoides tridactylus*	TTWO	Resident	Yes	0.04 ± 0.03	0	0	0.64 ± 0.63	0
Northern Flicker	*Colaptes auratus*	NOFL	Migrant	Yes	0.13 ± 0.05	0	0	0	0.64 ± 0.63
Olive-sided Flycatcher	*Contopus borealis*	OSFL	Migrant	Yes	0	0.64 ± 0.63	0	0	0
Alder Flycatcher	*Empidonax alnorum*	ALFL	Migrant	No	0.02 ± 0.02	0	0	0	0
Least Flycatcher	*Empidonax minimus*	LEFL	Migrant	Yes	0.02 ± 0.02	0	0	0	0
Tree Swallow	*Tachycineta bicolor*	TRSW	Migrant	Yes	0.04 ± 0.03	0	0	0.64 ± 0.63	0
Gray Jay	*Perisoreus canadensis*	GRJA	Resident	Yes	0.42 ± 0.15	1.91 ± 1.27	0	0	0
Black-capped Chickadee	*Parus atricapillus*	BCCH	Resident	No	1.13 ± 0.16	0.64 ± 0.63	0	0	0
Boreal Chickadee	*Parus hudsonicus*	BOCH	Resident	No	0.79 ± 0.13	0	0	0	0
Red-breasted Nuthatch	*Sitta canadensis*	RBNU	Resident	Yes	0.35 ± 0.10	0	0.64 ± 0.63	0	0
Brown Creeper	*Certhia americana*	BRCR	Migrant	Yes	0.06 ± 0.04	0	0	0	0
Winter Wren	*Troglodytes troglodytes*	WIWR	Migrant	Yes	0.02 ± 0.02	0	0	0	0
Golden-crowned Kinglet	*Regulus satrapa*	GCKI	Migrant	Yes	0.96 ± 0.14	0	0	0	0
Ruby-crowned Kinglet	*Regulus calendula*	RCKI	Migrant	Yes	2.60 ± 0.22	1.91 ± 1.27	0	0	0
Swainson's Thrush	*Catharus ustulatus*	SWTH	Migrant	No	0.77 ± 0.13	1.27 ± 1.27	0	0	0
Hermit Thrush	*Catharus guttatus*	HETH	Migrant	No	0.40 ± 0.09	0.64 ± 0.63	0	0	0
American Robin	*Turdus migratoruis*	AMRO	Migrant	No	0.04 ± 0.03	2.54 ± 0.63	0	1.91 ± 1.27	0
Cedar Waxwing	*Bombycilla cedrorum*	CEWA	Migrant	No	0.04 ± 0.03	0	0	0	0
Solitary Vireo	*Vireo solitarius*	SOVI	Migrant	Yes	0.44 ± 0.10	0	0	0	0
Warbling Vireo	*Vireo gilvus*	WAVI	Migrant	Yes	0.15 ± 0.05	0.64 ± 0.63	0.64 ± 0.63	0	0
Red-eyed Vireo	*Vireo olivaceus*	REVI	Migrant	Yes	0.08 ± 0.04	0	0	0	0
Orange-crowned Warbler	*Vermivora celata*	OCWA	Migrant	No	0.06 ± 0.04	0	0	0	0

TABLE 7.1. cont.

Common Name	Genus And Species	Species Code	Migration Habits	Use Large Trees	Density (number/10 ha)				
					Pre-Harvest Sites	Large Clumps	Medium Clumps	Small Clumps	Cutover Sites
Yellow Warbler	*Dendroica petechia*	YEWA	Migrant	No	0.29 ± 0.08	0	0	0	0
American Redstart	*Setophaga ruticilla*	AMRE	Migrant	No	0.04 ± 0.03	0	0	0	0
Ovenbird	*Seiurus aurocapillus*	OVEN	Migrant	No	0.06 ± 0.04	0	0	0	0
Mourning Warbler	*Oporornis philadelphia*	MOWA	Migrant	No	0.08 ± 0.04	0	0	0	0
Common Yellowthroat	*Geothlypis trichas*	COYE	Migrant	No	0.02 ± 0.02	0	0	0	0
Canada Warbler	*Wilsonia canadensis*	CAWA	Migrant	No	0.06 ± 0.04	0	0	0	0
Rose-breasted Grosbeak	*Pheucticus ludovicianus*	RBGR	Migrant	No	0.02 ± 0.02	0	0	0	0
Chipping Sparrow	*Spizella passerina*	CHSP	Migrant	Yes	1.58 ± 0.19	1.91 ± 0.78	1.27 ± 0.78	0.64 ± 0.63	1.91 ± 1.27
Lincoln's Sparrow	*Melospiza lincolnii*	LISP	Migrant	No	0.02 ± 0.02	1.27 ± 0.78	3.17 ± 1.00	3.81 ± 1.85	1.91 ± 1.27
White-throated Sparrow	*Zonotrichia albicollis*	WTSP	Migrant	No	0.69 ± 0.14	6.35 ± 2.84	3.81 ± 2.33	2.54 ± 1.19	0.64 ± 0.63
White-crowned Sparrow	*Zonotrichia leucophrys*	WCSP	Migrant	No	0	0.64 ± 0.63	0	0	0
Dark-eyed Junco	*Junco hyemalis*	DEJU	Migrant	No	1.23 ± 0.17	2.54 ± 1.19	7.62 ± 1.62	3.81 ± 1.19	5.08 ± 3.27
Brown-headed Cowbird	*Molothrus ater*	BHCO	Migrant	No	0.06 ± 0.04	0	0	0	0
White-winged Crossbill	*Loxia leucoptera*	WWCR	Resident	Yes	0.02 ± 0.02	0	0	0	0
Pine Siskin	*Carduelis pinus*	PISI	Migrant	Yes	0.29 ± 0.09	1.27 ± 1.27	1.91 ± 1.91	2.54 ± 2.54	0

TABLE 7.2. Mean (± SE) species richness and density per ha within each stand group and treatment type in the Whitecourt study area pre-harvest. Species were categorized by whether or not they used large trees or snags and whether or not they migrated. There were 3 and 121 degrees of freedom for the tests involving stand groups and 2 and 119 degrees of freedom for the tests involving treatment types. Groups/treatments with different superscripts are statistically different (Student-Neuman-Keuls range test).

Habitat Categories	Use Large Trees		Do Not Use Large Trees		All Categories Combined
	Resident	Migrant	Resident	Migrant	
Bird Species Richness					
Stand Group A	0.2 ± 0.1^{b}	2.9 ± 0.2	0.6 ± 0.1	1.1 ± 0.1	4.8 ± 0.3^{ab}
Stand Group B	0.5 ± 0.1^{a}	3.0 ± 0.2	0.6 ± 0.1	1.5 ± 0.2	5.6 ± 0.4^{a}
Stand Group C	0.2 ± 0.1^{b}	2.8 ± 0.2	0.7 ± 0.1	1.5 ± 0.2	5.1 ± 0.3^{ab}
Stand Group D	0.2 ± 0.1^{b}	2.3 ± 0.3	0.6 ± 0.1	1.4 ± 0.2	4.5 ± 0.4^{b}
Test Statistic (F)	3.5	1.6	1.6	0.9	2.49
Probability Level (P)	0.02	0.19	0.22	0.49	0.05
Treatment 1	0.2 ± 0.1	2.5 ± 0.2	0.8 ± 0.1	1.2 ± 0.2	4.7 ± 0.3
Treatment 2	0.3 ± 0.1	2.9 ± 0.2	0.5 ± 0.1	1.5 ± 0.2	5.3 ± 0.3
Treatment 3	0.3 ± 0.1	2.8 ± 0.2	0.6 ± 0.1	1.4 ± 0.2	5.0 ± 0.3
Test Statistic (F)	0.6	0.8	0.3	1.2	1.3
Probability Level (P)	0.94	0.43	0.77	0.32	0.27
Total Bird Density					
Stand Group A	0.4 ± 0.2	3.3 ± 0.2	0.6 ± 0.1	1.2 ± 0.1	5.5 ± 0.4
Stand Group B	0.6 ± 0.1	3.5 ± 0.3	0.7 ± 0.1	1.6 ± 0.2	6.3 ± 0.5
Stand Group C	0.3 ± 0.2	3.2 ± 0.3	0.7 ± 0.1	1.6 ± 0.2	5.9 ± 0.4
Stand Group D	0.2 ± 0.1	2.7 ± 0.3	0.7 ± 0.1	1.5 ± 0.2	5.1 ± 0.4
Test Statistic (F)	1.2	1.8	0.1	0.8	2.3
Probability Level (P)	0.31	0.14	0.95	0.46	0.08
Treatment 1	0.3 ± 0.1	2.9 ± 0.2	0.8 ± 0.1	1.2 ± 0.1	5.2 ± 0.4
Treatment 2	0.5 ± 0.1	3.4 ± 0.3	0.5 ± 0.1	1.7 ± 0.2	6.2 ± 0.5
Treatment 3	0.3 ± 0.1	3.2 ± 0.2	0.7 ± 0.1	1.4 ± 0.2	5.6 ± 0.4
Test Statistic (F)	0.4	1.3	0.5	1.7	2.0
Probability Level (P)	0.65	0.29	0.61	0.19	0.11

iterations, ter Braak 1992). After adjusting for differences among stand groups, bird communities did not differ among treatment types pre-harvest ($F = 1.2$, $P = 0.10$; dummy variables coding both stand groups and treatment types were included in the CCA, and permutations during the bootstrap Monte Carlo test were restricted to within the appropriate stand group, ter Braak 1992).

Post-harvest - One hundred and sixty-two individuals were detected within the 15 residual clumps and 5 cutover sites that were surveyed for birds during 1998. Only 18 of the bird species detected pre-harvest (43%) were detected post-harvest (Table 7.1) and three new species (Killdeer, Olive-sided Flycatcher, White-crowned Sparrow) were detected. Some of the species that were not detected post-harvest may have been present in the harvest area but overlooked because fewer sites were surveyed post-harvest than pre-harvest. In fact, the proportion of species that were resident was similar pre- and post-harvest (23% and 19% of the species, respectively, $\chi^2 = 0.2$, $P = 0.66$, $df = 1$), and the proportion of species that nest or forage in large trees or snags was similar pre- and post-harvest (52% and 52% of the species, respectively, $\chi^2 = 0.0$, $P > 0.99$, $df = 1$).

For both resident and migrant birds that nest or forage in large trees, for migrant birds that do not nest or forage in large trees, and for all birds combined, total density of birds differed among residual clump sizes (Table 7.3). Densities were not adjusted for differences between stand groups in these analyses because only two stand groups were surveyed during 1998. Large residual clumps had higher densities of birds that use large trees than did medium and small residual clumps, or than did cutover sites. In addition large residual clumps tended to have higher densities of birds that use large trees than did pre-harvest sites, but that difference was not statistically significant (Table 7.3). Resident birds that do not use large trees had higher densities in pre-harvest than in any of the post-harvest sites, whereas migrant species that do not use large trees had higher densities in all post-harvest sites than they had pre-harvest (Table 7.3).

To ensure that the data would not be heavily biased towards pre-harvest sites in the CAs and CCAs, only 12 pre-harvest sites (one randomly chosen from each of the stands) were included. The first and second functions of the CA had eigen values of 0.55 and 0.32, respectively. Combined, these two functions accounted for 22% of the variation in bird species density. Bird communities differed among the five habitat types (Fig. 7.1; $F = 2.3$, $P < 0.001$; dummy variables coding each habitat type were included as environmental variables in the CCA, and differences among habitat types were tested using a bootstrap Monte Carlo test with 999 iterations, ter Braak 1992). Bird communities in cutover sites, however, did not differ from those in small residual clumps ($F = 1.1$, $P = 0.40$), which in turn did not differ from those in medium residual clumps ($F = 1.1$, $P = 0.45$). Bird communities in cutover sites, small residual clumps, and medium residual clumps differed from those in large residual clumps ($F = 2.9$, $P < 0.001$), which in turn differed from those in pre-harvest sites ($F = 1.7$, $P < 0.001$).

All except two of the resident bird species that were included in the correspondence analyses (i.e., bird species that were detected on three or more occasions) had higher predicted densities in pre- than post-harvest sites (Fig. 7.2; bird species are plotted on the ordination diagram at their highest predicted density; ter Braak 1992). The remaining two resident species had their highest predicted densities within large residual clumps. For migrating bird species, however, only 60% of the species had higher predicted densities in pre- than in post-harvest sites. For both bird species that use large trees and for those that do not use large trees, about 60% had higher predicted densities in pre- than in post-harvest sites (Fig. 7.2).

Grande Prairie Study Area

Pre-harvest - Within the 12 stands (120 sites) that were surveyed pre-harvest, 50 bird species and 1592 individuals were detected (Table 7.4). This bird community was dominated by species that migrate with only 9 species (18%) being year-round residents. Most residents (67% of 9 species), but fewer migrants (34% of 41 species) nested or foraged in large live or dead trees, but

TABLE 7.3. Density of birds per 10 ha (24.7 acre) (mean ± SE) on the Whitecourt study area. Pre-harvest forest was surveyed during 1997, clumps and cutover areas were surveyed during 1998. Species were categorized based on whether or not they used large trees or snags for nesting and foraging and whether or not they migrated. Habitats with different superscripts have statistically different densities (Student-Neuman-Keuls range test).

Habitat	Number Of Sites Surveyed	Use Large Trees		Do Not Use Large Trees		Total
		Resident	Migrant	Resident	Migrant	
Pre-Harvest Sites	120	1.0 ± 0.2	11.1 ± 0.5^{a}	1.9 ± 0.2^{a}	4.0 ± 0.3^{c}	17.9 ± 0.6^{b}
Large Residual Clumps	5	1.9 ± 1.3	12.7 ± 4.3^{a}	0.6 ± 06^{b}	18.4 ± 6.8^{a}	33.7 ± 11.2^{a}
Medium Residual Clumps	5	0.6 ± 0.6	3.8 ± 3.1^{b}	0 ± 0^{b}	17.1 ± 4.8^{a}	21.6 ± 5.3^{b}
Small Residual Clumps	5	0.6 ± 0.6	4.4 ± 2.9^{b}	0 ± 0^{b}	14.6 ± 2.2^{ab}	19.7 ± 0.4^{b}
Cutover Sites	5	0 ± 0	2.5 ± 1.9^{b}	0 ± 0^{b}	9.5 ± 2.2^{b}	12.1 ± 3.1^{b}
Test Statistic F		0.6	6.8	3.2	26.5	5.3
Probability Level (P)		0.66	<0.001	0.01	<0.001	<0.001

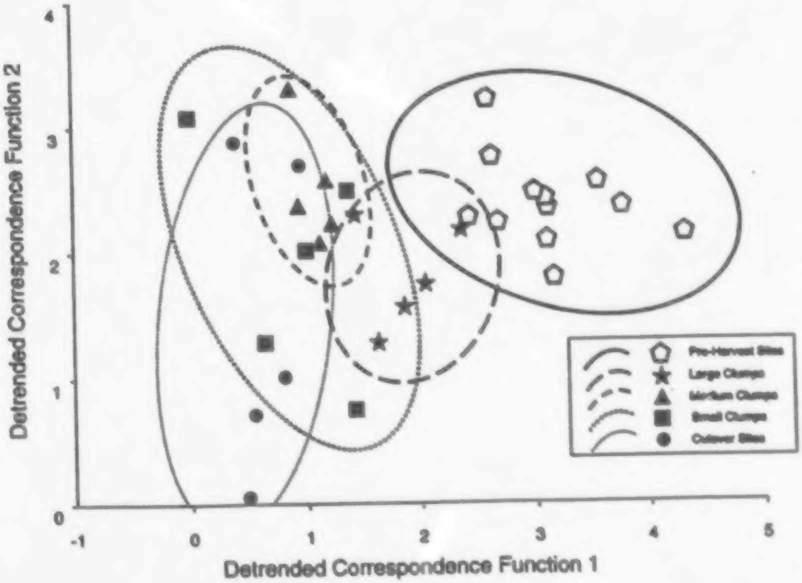


FIGURE 7.1. Correspondence analyses based on bird densities in pre- and post-harvest sites near Whitecourt AB during 1998. Harvest sites were categorized based on presence and size of residual tree clumps. Sites that are close together within the correspondence plot have similar bird communities and sites far apart have dis-similar bird communities. The bivariate ellipse for each habitat type encloses 90% of the sites (Systat Inc. 1989).

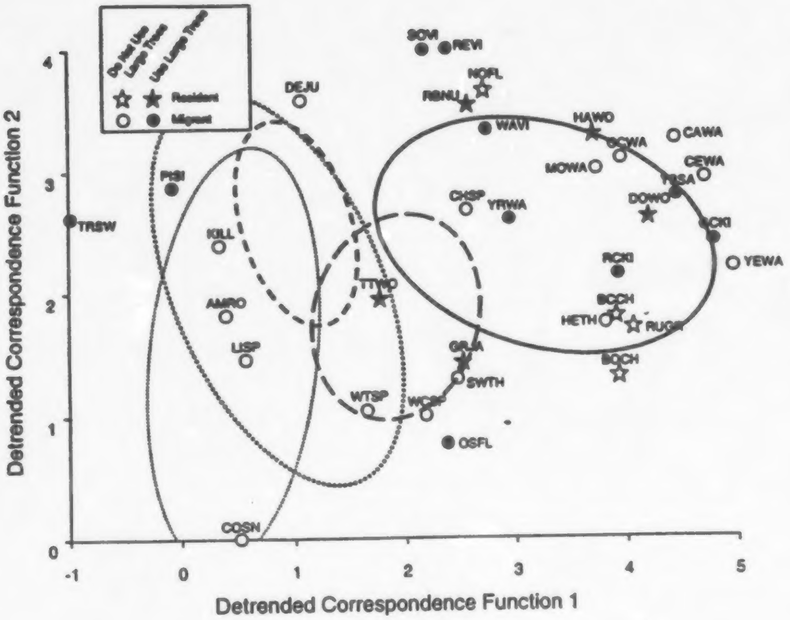


FIGURE 7.2. Predicted location of maximum density for each bird species based on the correspondence analyses of bird densities in pre- and post-harvest forest sites near Whitecourt, AB during 1998. Species that are close together have strong positive covariance in density whereas species that are far apart have strong negative covariance. Species were categorized based on their migration and nesting/foraging habits. Bivariate ellipses from Fig. 7.1 are included for reference.

that difference was not statistically significant ($\chi^2 = 3.3$, $P = 0.07$, $df = 1$).

Since vegetation in and around the stands differed among stand groups pre-harvest (Chapters 2-5), I evaluated whether it was necessary to control for differences among stand groups before determining whether density or species richness differed among treatments. Both density of birds and species richness differed among stand groups (Table 7.5). Differences in species richness among stand groups were mainly due to differences for birds that did not use large trees and snags. Differences in density among stand groups were present for all groups except resident species that use large trees and snags. Thus, I adjusted for differences among stand groups before evaluating whether differences occurred among treatment types. Within this analysis, bird density and species richness were similar among treatments for each of the categories of birds and when all categories were combined (Tables 7.5).

Similar results were found within the community analyses. The first and second functions of the CA had eigen values of 0.07 and 0.04, respectively. Combined, these two functions accounted for only 4.1% of the variation in bird species density. However, even with this small amount of explained variation, bird communities differed among stand groups pre-harvest ($F = 1.9$, $P < 0.001$). After adjusting for differences among stand groups, bird communities did not differ among treatment types pre-harvest ($F = 1.2$, $P = 0.14$).

TABLE 7.4. Species names, codes, migration habits, use of large trees for nesting or foraging, and density (mean ± SE) within pre- and post-harvest sites in the Grande Prairie study area, that were surveyed during 1997 and 1998.

Common Name	Genus And Species	Species Code	Migration Habits	Use Large Trees	Density (number/10 ha) Pre-Harvest Sites	Large Clumps	Medium Clumps	Small Clumps	Cutover Sites
Ruffed Grouse	*Bonasa umbellus*	RUGR	Resident	No	0.08 ± 0.04	0	0	0	0
Kildeer	*Charadrius vociferus*	KILL	Migrant	No	0	0	0	0.64 ± 0.63	0
Common Snipe	*Gallinago gallinago*	COSN	Migrant	No	0.02 ± 0.02	0.64 ± 0.63	1.91 ± 0.78	0.64 ± 0.63	0.64 ± 0.63
Yellow-bellied Sapsucker	*Sphyrapicus varius*	YBSA	Migrant	Yes	0.98 ± 0.14	0	0	0	0
Downy Woodpecker	*Picoides pubescens*	DOWO	Resident	Yes	0.04 ± 0.04	0.64 ± 0.63	0	0	0
Hairy Woodpecker	*Picoides villosus*	HAWO	Resident	Yes	0.10 ± 0.05	0.64 ± 0.63	0	0	0
Northern Flicker	*Colaptes auratus*	NOFL	Migrant	Yes	0.19 ± 0.08	0	1.27 ± 0.78	0	0
Pileated Woodpecker	*Dryocopus pileatus*	PIWO	Resident	Yes	0.02 ± 0.02	0	0	0	0
Western Wood-pewee	*Contopus sordidulus*	WWPE	Migrant	No	0	1.27 ± 0.78	0	0	0
Alder Flycatcher	*Empidonax alnorum*	ALFL	Migrant	No	0.21 ± 0.06	0	0	0	0
Least Flycatcher	*Empidonax minimus*	LEFL	Migrant	Yes	2.98 ± 0.37	8.26 ± 4.21	0	0	0
Tree Swallow	*Tachycineta bicolor*	TRSW	Migrant	Yes	0.04 ± 0.04	0	0	0	0
Gray Jay	*Perisoreus canadensis*	GRJA	Resident	Yes	0.06 ± 0.06	0	0	0	0
Common Raven	*Corvus corax*	CORA	Resident	Yes	0.06 ± 0.05	0	0	0	0
Black-capped Chickadee	*Parus atricapillus*	BCCH	Resident	No	0.73 ± 0.15	0.64 ± 0.63	0	0	0
Boreal Chickadee	*Parus hudsonicus*	BOCH	Resident	No	0.06 ± 0.04	0	0	0	0
Red-breasted Nuthatch	*Sitta canadensis*	RBNU	Resident	Yes	0.15 ± 0.05	0	0	0	0
Winter Wren	*Troglodytes troglodytes*	WIWR	Migrant	Yes	0	0.64 ± 0.63	0	0	0
Golden-crowned Kinglet	*Regulus satrapa*	GCKI	Migrant	Yes	0.02 ± 0.02	0	0.64 ± 0.63	0	0
Ruby-crowned Kinglet	*Regulus calendula*	RCKI	Migrant	Yes	0.10 ± 0.05	0	0	0	0
Veery	*Catharus fuscescens*	VEER	Migrant	No	0.02 ± 0.02	0	0	0	0
Swainson's Thrush	*Catharus ustulatus*	SWTH	Migrant	No	0.79 ± 0.12	1.91 ± 1.27	0	0	0
Hermit Thrush	*Catharus guttatus*	HETH	Migrant	No	0.31 ± 0.09	1.27 ± 0.78	0	0	0
American Robin	*Turdus migratoruis*	AMRO	Migrant	No	0.46 ± 0.11	5.08 ± 0.78	2.54 ± 1.19	0	0
Cedar Waxwing	*Bombycilla cedrorum*	CEWA	Migrant	No	0.06 ± 0.04	0	0	0	0
Solitary Vireo	*Vireo solitarius*	SOVI	Migrant	Yes	0.06 ± 0.04	0	0	0	0
Warbling Vireo	*Vireo gilvus*	WAVI	Migrant	Yes	0.31 ± 0.08	6.99 ± 3.54	1.27 ± 0.78	0	0
Philadelphia Vireo	*Vireo philadelphicus*	PHVI	Migrant	Yes	0	0.64 ± 0.63	0	0	0
Red-eyed Vireo	*Vireo olivaceus*	REVI	Migrant	Yes	1.06 ± 0.14	1.27 ± 0.78	0	0	0
Tennessee Warbler	*Vermivora peregrina*	TEWA	Migrant	No	0.85 ± 0.13	0	0	0	0

FIGURE 7.4. cont.

Common Name	Genus And Species	Species Code	Migration Habits	Use Large Trees	Density (number/10 ha)				
					Pre-Harvest Sites	Large Clumps	Medium Clumps	Small Clumps	Cutover Sites
Orange-crowned Warbler	*Vermivora celata*	OCWA	Migrant	No	0.15 ± 0.05	0	0	0	0
Connecticut Warbler	*Oporornis agilis*	COWA	Migrant	No	0.27 ± 0.07	0	0	0	0
Yellow Warbler	*Dendroica petechia*	YEWA	Migrant	No	1.21 ± 0.18	0	0	0	0
Magnolia Warbler	*Dendroica magnolia*	MAWA	Migrant	No	0.15 ± 0.06	0	0	0	0
Yellow-rumped Warbler	*Dendroica coronata*	YRWA	Migrant	Yes	3.88 ± 0.28	2.54 ± 1.19	0.64 ± 0.63	0	1.27 ± 1.27
Black-throated Green Warbler	*Dendroica virens*	BTGW	Migrant	Yes	0.08 ± 0.05	0	0	0	0
Blackpoll Warbler	*Dendroica striata*	BPWA	Migrant	No	0.02 ± 0.02	0	0	0	0
Black and white Warbler	*Mniotilta varia*	BWWA	Migrant	No	0.02 ± 0.02	0.64 ± 0.63	0	0	0
American Redstart	*Setophaga ruticilla*	AMRE	Migrant	No	1.31 ± 0.18	3.81 ± 2.33	0	0	0
Ovenbird	*Seiurus aurocapillus*	OVEN	Migrant	No	1.81 ± 0.21	1.27 ± 1.27	0	0	0
Northern Waterthrush	*Sieurus noveboracensis*	NOWA	Migrant	No	0.08 ± 0.04	0	0	0	0
Mourning Warbler	*Oporornis philadelphia*	MOWA	Migrant	No	0.17 ± 0.06	1.27 ± 0.78	1.91 ± 0.78	0	0
Common Yellowthroat	*Geothlypis trichas*	COYE	Migrant	No	0.04 ± 0.03	0	0	0	0
Canada Warbler	*Wilsonia canadensis*	CAWA	Migrant	No	0.02 ± 0.02	0	0	0	0
Western Tanager	*Piranga ludoviciana*	WETA	Migrant	Yes	0.63 ± 0.13	0.64 ± 0.63	0.64 ± 0.63	0.64 ± 0.63	0
Rose-breasted Grosbeak	*Pheucticus ludovicianus*	RBGR	Migrant	No	1.19 ± 0.19	1.27 ± 0.78	0	0	0
Chipping Sparrow	*Spizella passerina*	CHSP	Migrant	Yes	1.58 ± 0.22	1.27 ± 0.78	1.27 ± 0.78	0	0.64 ± 0.63
Song Sparrow	*Melospiza melodia*	SOSP	Migrant	No	0.04 ± 0.03	0	0	0	0
Lincoln's Sparrow	*Melospiza lincolnii*	LISP	Migrant	No	0.56 ± 0.13	1.91 ± 1.91	5.71 ± 2.11	1.91 ± 0.78	3.17 ± 1.42
White-throated Sparrow	*Zonotrichia albicollis*	WTSP	Migrant	No	7.44 ± 0.43	6.99 ± 1.85	8.26 ± 1.27	5.08 ± 1.62	2.54 ± 1.19
Dark-eyed Junco	*Junco hyemalis*	DEJU	Migrant	No	1.31 ± 0.18	0.64 ± 0.63	2.54 ± 1.19	1.91 ± 1.27	3.17 ± 1.42
Red-winged Blackbird	*Agelaius phoeniceus*	RWBL	Migrant	No	0.08 ± 0.05	1.27 ± 0.78	0	0	0
Rusty Blackbird	*Euphagus carolinus*	RUBL	Migrant	No	0	0.64 ± 0.63	0	0	0
Brown-headed Cowbird	*Molothrus ater*	BHCO	Migrant	No	0.83 ± 0.17	0	0.64 ± 0.63	0	0.64 ± 0.63
Pine Siskin	*Carduelis pinus*	PISI	Migrant	Yes	0.27 ± 0.13	0	0	0	0

TABLE 7.5. Mean (± SE) species richness and density per ha within each stand group and treatment type in the Grande Prairie study area pre-harvest. Species were categorized by whether or not they used large trees or snags and whether or not they migrated. There were 3 and 121 degrees of freedom for the tests involving stand groups and 2 and 119 degrees of freedom for the tests involving treatment types. Groups/treatments with different superscripts are statistically different (Student-Neuman-Keuls range test).

Habitat Categories	Use Large Trees		Do Not Use Large Trees		All Categories Combined
	Resident	Migrant	Resident	Migrant	
Bird Species Richness					
Stand Group A	0.1 ± 0.1	3.2 ± 0.2	0.3 ± 0.1^{ab}	4.1 ± 0.2^{b}	7.7 ± 0.4^{b}
Stand Group B	0.1 ± 0.1	2.7 ± 0.2	0.4 ± 0.1^{a}	4.0 ± 0.2^{b}	7.3 ± 0.4^{b}
Stand Group C	0.2 ± 0.1	3.4 ± 0.2	0.1 ± 0.1^{b}	5.3 ± 0.3^{a}	9.0 ± 0.5^{a}
Stand Group D	0.1 ± 0.1	2.8 ± 0.2	0.3 ± 0.1^{ab}	5.0 ± 0.3^{a}	8.2 ± 0.4^{a}
Test Statistic (F)	0.2	2.5	3.2	6.4	4.6
Probability Level (P)	0.92	0.06	0.03	0.001	0.001
Treatment 1	0.1 ± 0.1	3.0 ± 0.2	0.3 ± 0.1	4.5 ± 0.2	7.9 ± 0.04
Treatment 2	0.1 ± 0.1	3.2 ± 0.2	0.2 ± 0.1	5.0 ± 0.3	8.5 ± 0.06
Treatment 3	0.1 ± 0.1	2.9 ± 0.2	0.3 ± 0.1	4.3 ± 0.3	7.6 ± 0.04
Test Statistic (F)	0.6	0.8	0.3	2.3	2.2
Probability Level (P)	0.94	0.44	0.77	0.10	0.14
Total Bird Density					
Stand Group A	0.1 ± 0.1	3.9 ± 0.3^{ab}	0.3 ± 0.1^{ab}	5.0 ± 0.3^{c}	9.3 ± 0.5^{b}
Stand Group B	0.1 ± 0.1	3.4 ± 0.3^{ab}	0.6 ± 0.2^{a}	4.7 ± 0.3^{c}	8.8 ± 0.4^{b}
Stand Group C	0.2 ± 0.1	4.4 ± 0.4^{a}	0.1 ± 0.1^{b}	7.4 ± 0.5^{a}	12.0 ± 0.6^{a}
Stand Group D	0.2 ± 0.1	3.1 ± 0.2^{b}	0.3 ± 0.1^{ab}	6.2 ± 0.4^{b}	9.8 ± 0.5^{b}
Test Statistic (F)	0.3	3.3	3.2	11.8	9.7
Probability Level (P)	0.86	0.02	0.03	<0.001	<0.001
Treatment 1	0.1 ± 0.1	3.5 ± 0.2	0.3 ± 0.1	5.6 ± 0.2	9.6 ± 0.5
Treatment 2	0.1 ± 0.1	4.3 ± 0.4	0.3 ± 0.1	6.2 ± 0.3	10.7 ± 0.6
Treatment 3	0.2 ± 0.1	3.3 ± 0.3	0.3 ± 0.1	5.5 ± 0.4	9.6 ± 0.5
Test Statistic (F)	0.4	2.5	0.1	1.5	2.0
Probability Level (P)	0.66	0.09	0.87	0.22	0.11

Post-harvest - One hundred and sixty-nine individuals were detected in the 15 residual clumps and 5 cutover sites that were surveyed for birds post-harvest (Table 7.4). Only 25 of the bird species (50%) that were detected pre-harvest were also detected within the residual clumps or cutover sites post-harvest (Table 7.4). Five new species (Killdeer, Western Wood-Peewee, Winter Wren, Warbling Vireo, Rusty Blackbird) were also detected post-harvest. Some of the species that were not detected post-harvest may have been present but overlooked because few post-harvest sites were surveyed. In fact the proportion of species that were residents was similar pre- and post-harvest (18% and 10% of the species, respectively; $\chi^2 = 0.9$, $P = 0.33$, df = 1) and the proportion of species nesting and/or foraging in large live trees or snags was similar pre- and post-harvest (38% and 36% of the species, respectively; $\chi^2 = 0.1$, $P = 0.81$, df = 1).

For migrant birds that use large trees, migrant birds that do not use large trees, and all birds combined, density differed among habitat types post-harvest (Table 7.6). Densities were not adjusted for differences between stand groups in these analyses because only two stand groups were surveyed during 1998. Large residual clumps had higher densities of migrant birds than did medium residual clumps or pre-harvest sites, which in turn had higher densities than did small residual clumps or cutover sites (Table 7.6). Resident birds were present at low densities in pre-harvest sites and in large residual clumps, and absent in smaller residual clumps and in cutover sites, but those differences were not statistically significant (Table 7.6).

To ensure that the data used in the correspondence analyses would not be heavily biased towards pre-harvest sites, only 12 pre-harvest sites (one randomly chosen from each of the stands) were included. The first and second functions of the CA had eigen values of 0.47 and 0.34, respectively, and combined, these two functions accounted for 27% of the variation in bird species density. Bird communities differed among the five habitat types (Fig. 7.3; $F = 2.1$, $P < 0.001$). Bird communities in cutover sites, however, did not differ from those in small residual clumps ($F = 0.7$, $P = 0.75$), which in turn did not differ from those in medium residual clumps ($F = 1.1$, $P = 0.30$). Bird communities in cutover sites, small residual clumps, and medium residual clumps differed from those in large residual clumps ($F = 3.4$, $P < 0.001$), which in turn differed from those in pre-harvest forests ($F = 2.4$, $P < 0.001$).

All except two of the resident bird species that were included in the correspondence had higher predicted densities in pre- than in post-harvest sites (Fig. 7.4). The remaining two resident species had their highest predicted densities within large residual clumps. For migrating bird species, however, only 45% had higher predicted densities in pre- than in post-harvest sites. About 30% of the migrating species had higher predicted density in large residual clumps than in pre-harvest sites, smaller residual clumps, or cutover sites. About 60% of the bird species that use large trees had higher predicted densities in pre- than in post-harvest sites, with all but one of the remaining species that use large trees having highest predicted densities in large residual clumps (Fig. 7.4). Only about 40% of the bird species that do not use large trees had higher predicted densities in pre- than in post-harvest sites (Fig. 7.4).

Discussion

Pre-harvest

Within both study areas, some of the variation among bird communities was related to differences among stand groups. However, that variation was small and the bird communities overlapped greatly among the stand groups. After controlling for the differences among stand groups, bird communities did not differ among treatment types pre-harvest. Consequently, differences we found in bird communities post-harvest can be attributed unambiguously to differences in the size and dispersion of tree clumps that were retained in the experimental areas. Thus, this study provides a rigorous comparison of bird communities among harvest areas where three patterns of residual trees were retained.

TABLE 7.6. Density of birds per 10 ha (24.7 acre) (mean ± SE) on the Grande Prairie study area. Pre-harvest forest was surveyed during 1997, clumps and cutover areas were surveyed during 1998. Species were categorized based on whether or not they used large trees or snags and whether or not they migrated. Habitats with different superscripts have statistically different densities (Student-Neuman-Keuls range test).

Habitat	Number Of Sites Surveyed	Use Large Trees		Do Not Use Large Trees		Total
		Resident	Migrant	Resident	Migrant	
Pre-Harvest Forest Sites	120	0.4 ± 0.1	12.2 ± 0.6^{b}	0.9 ± 0.2	19.4 ± 0.7^{bc}	32.9 ± 1.0^{b}
Large Residual Clumps	5	1.3 ± 1.3	21.6 ± 7.4^{a}	0.6 ± 06	31.1 ± 4.9^{a}	54.6 ± 9.1^{a}
Medium Residual Clumps	5	0 ± 0	5.7 ± 1.9^{bc}	0 ± 0	24.1 ± 3.4^{ab}	29.8 ± 5.2^{b}
Small Residual Clumps	5	0 ± 0	0.6 ± 0.6^{c}	0 ± 0	10.2 ± 1.9^{c}	10.8 ± 1.6^{c}
Cutover Sites	5	0 ± 0	1.9 ± 1.3^{c}	0 ± 0	10.2 ± 3.2^{c}	12.1 ± 4.2^{c}
Test Statistic F		0.9	9.7	0.9	6.4	13.4
Probability Level (P)		0.48	<0.001	0.48	<0.001	<0.001

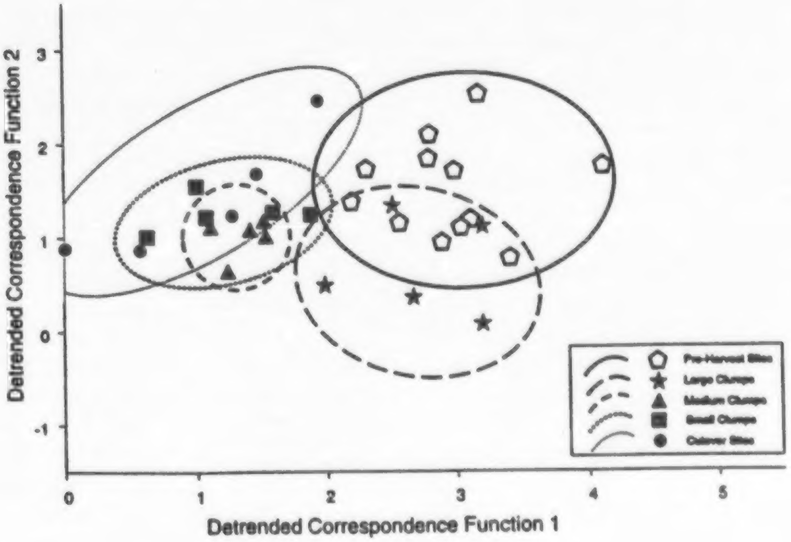


FIGURE 7.3. Correspondence analyses based on bird densities in pre- and post-harvest sites near Grande Prairie, AB during 1998. Harvest sites were categorized based on presence and size of residual tree clumps. Sites that are close together within the correspondence plot have similar bird communities and sites far apart have dis-similar bird communities. The bivariate ellipse for each habitat type encloses 90% of the sites.

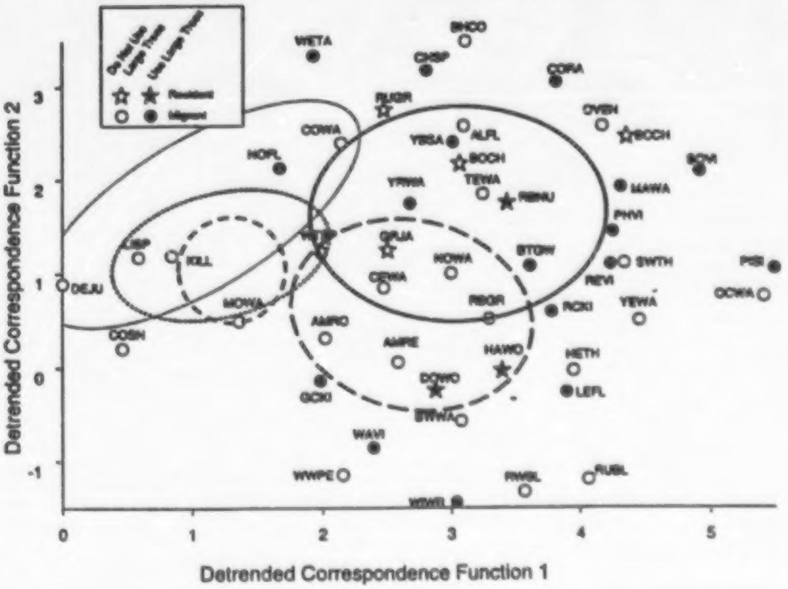


FIGURE 7.4. Predicted location of maximum density for each bird species based on the correspondence analyses of bird densities in pre- and post-harvest forest sites near Grande Prairie, AB during 1998. Species that are close together have strong positive covariance in density whereas species that are far apart have strong negative covariance. Species were categorized based on their migration and nesting/foraging habits. Bivariate ellipses from Fig. 7.3 are included for reference.

Post-harvest

Due to funding considerations, only cursory bird surveys were conducted during the 1998 field season. Consequently, only a limited analyses and interpretation of bird communities post-harvest will be presented in this report. A more detailed analyses will be presented in the final report March 2000.

In both study areas, fewer birds were detected in cutover sites than in the pre-harvest sites. Thus, some of the birds that lived in the cut-over areas pre-harvest did not return post-harvest. However, more birds were detected in large residual clumps than in the pre-harvest forests. A relatively high density of birds in large residual clumps, and low density in cutover sites, may indicate that some of the birds that lived in the stands pre-harvest moved into the large residual clumps post-harvest. A similar pattern has been detected in other studies (Lovejoy et al. 1986, Schmiegelow et al. 1997). Density of birds in large residual clumps may decrease during the next few years as some of the birds that are present there die, and others move to less crowded habitats (Schmiegelow et al. 1997). I will compare bird density during 1998 with that found during 1999 to evaluate whether bird density decreases in residual clumps between the first and second years post-harvest.

The composition of the bird community differed among residual clump sizes. Species typically associated with open country (e.g., Killdeer, Lincoln's Sparrow,) and shrubby areas (e.g., Morning Warbler, White-crowned Sparrow) had higher densities within cutover sites, and within small and medium residual clumps, than they had within large residual clumps or within pre-harvest sites. Conversely, birds typically associated with large trees (e.g., Hairy Woodpecker, Red-breasted Nuthatch) and closed canopy forests (e.g., Swainson's Thrush, Yellow-rumped Warbler) had higher densities in large residual clumps and pre-harvest sites than they had in medium and small residual clumps. Those patterns were statistically significant and consistent for both study areas. Thus, within large residual clumps (i.e., clumps that had > 500 trees retained at harvest, Chapter 3), the bird community included many of the species that are associated with unharvested forest, whereas that was not true within medium and small residual tree clumps (i.e., clumps that had < 150 trees retained at harvest). Within the small and medium residual clumps, bird communities were more similar to those in cutover sites. Thus, there appeared to be a threshold between medium and large residual clumps where the bird community changed from being dominated by open country species to being dominated by forest species.

Hansen et al. (1995b) also found thresholds in tree density where forest bird abundance changed greatly. Within the large clumps we studied, bird communities were not identical to those in pre-harvest sites, but rather included some open country bird species as well as forest species. Thus, to retain bird communities that are similar to those found in unharvested forest it may be necessary to have clumps/reserves > 10 ha (24.7 acre) (Schmiegelow et al. 1997 and see below). Due to low sample size in the present analyses, variation in bird communities among clump sizes should be interpreted as preliminary. During 1999 intensity of bird surveys will be conducted and patterns will be re-evaluated at that time.

There are two potential explanations for large residual clumps and pre-harvest sites having bird communities that were more similar than were the bird communities between smaller residual clumps and pre-harvest sites. First, large residual clumps have more trees than smaller clumps, and as such may provide more resources for forest birds. Within in a variety of different forest types in North America (Dickson et al. 1983, Lambert 1992, Hansen et al. 1995a&b, MacKenzie and Steventon 1996, Annand and Thompson 1997, Norton and Hannon 1997, and Steventon et al. 1998) and in Tasmania (Taylor and Haseler 1995), there was a positive relationship between the density of trees retained at harvest and the similarity between bird communities in harvest and unharvested areas. The second potential explanation is that external environmental conditions (Chen et al. 1995) and biota (Laurance 1991, DellaSala and Rabe 1987, Saunders et al. 1991, Brothers and Spingarn 1992) may penetrate the edges of residual trees clumps, with the result that large residual clumps may retain more of the microhabitats that were present pre-harvest than do smaller residual clumps. As such, birds that require interior forest habitats may be present in large, but not in small, clumps (e.g., DellaSala and Rabe 1987).

The consequences to bird communities of retaining large versus small clumps of trees in cut-blocks has been evaluated only in the present study and in (Cameron 1996). In both studies, more individuals and more species were present in large than small residual clumps. In addition, within the present study, most of the birds that use large trees for nesting or foraging, and most resident species, had higher densities in large versus small residual clumps. Within large clumps in the present study, and in studies by Seip and Parker (1997) and Merrill et al. (1998), and in reserves of 1 or 10 ha (2.47 or 24.7 acre) (Schmiegelow et al. 1997) bird communities did not differ greatly from those found in unharvested forest. Thus, it appears that many forest birds have reasonable densities in large clumps.

At present there are few studies where the relationship between bird communities and clump size has been evaluated. Consequently, managers should not rely on residual tree clumps as the only habitat for forest birds (Hansen et al. 1991, Lindenmayer and Franklin 1997). In addition, the reproductive success of the birds in residual clumps has not been evaluated in any studies. Only after more extensive studies will we be able to determine whether retaining large clumps of residual trees in harvest sites provides viable populations of birds that are commonly associated with old forests. In the interim, it would be prudent to retain adequate amounts of unharvested forest in the landscape (Franklin 1993).

Bird communities in the present study, and most previous studies, were conducted in harvest areas that were less than five years old (but see Hansen et al. 1995a). Results from these studies should be interpreted with caution because the vegetation in cut-blocks is dynamic. As trees grow in cutover areas, biota and environmental conditions outside the clumps will become more similar to those within the clumps. Since edge effects probably decrease as the contrast between the vegetation inside and outside the clump decreases (DeGraaf 1992) small and medium clumps may have microhabitats more similar to large residual clumps by the time the new forest is 50-80 years old (Franklin 1993, Schieck et al. 1995). Thus, small or medium sized residual clumps spaced throughout a cut-block may allow biota that require old-growth microhabitats to use a larger proportion of the cut-block at mid rotation, than if all the trees had been retained in a few large clumps. Only by evaluating the relative merits of different clump sizes at various times throughout succession will managers be able to determine the relative merits of large versus small clumps. To that end the present study will be maintained throughout a rotation so changes in forest structures and biota can be re-surveyed periodically. Until results from these additional studies becomes available, however, managers may wish to include many clump sizes in harvest areas.

Although managers may wish to retain bird communities associated with unharvested forest in some cut-blocks, that goal may not be desirable in all harvest areas. Historically, wildfires were common in the boreal forest (Hunter 1993, Stelfox 1995) and those fires created a heterogeneous landscape with some areas having all trees killed and other areas having many residual live trees (Eberhart and Woodard 1987). To match that diversity of post-fire habitats, managers may wish to create some cut-blocks that have few or no residual trees, and other cut-blocks with clumps of residual trees.

Future Research

Only cursory bird surveys were conducted during 1998. Bird communities in post-harvest areas will be surveyed intensively during 1999. These intensive surveys will be conducted along transects that were established in each stand during 1997 (Figs. 2.1 and 2.2). In addition, birds will be surveyed using point counts at sites centered in residual clumps, cutover sites, and in adjacent unharvested forest sites. Twenty sites will be surveyed in cutover areas, small residual clumps, medium residual clumps and in unharvested forest adjacent to the experimental areas. Only eight and ten large clumps, in the Whitecourt and Grande Prairie study areas respectively, will be surveyed because that is all that are present (Fig. 2.1, 2.2). Small and medium residual clumps to be surveyed will be selected randomly with the restriction that an equal number of clumps will be surveyed from each stand group. Cutover sites will be > 100 m (109 yd.) from residual tree clumps. Sites in the unharvested forest will be > 200 m (219 yd.) from the forest edge. All sites and transects will be surveyed once during late April, and three times at bi-weekly intervals between 15 May and 30 June.

Literature cited

Alberta-Pacific Forest Industries Inc. 1996. An operator's guide to stand structure. Alberta-Pacific Forest Industries Inc., Boyle, AB.

Annand, E.M., Thompson, F.R. 1997. Forest bird responses to regeneration practices in central hardwood forests. J. Wild. Manage. 61: 159-171.

British Columbia Ministry of Forests. 1995. Biodiversity guidebook, forest practices code of British Columbia. British Columbia Ministry of Forests, Victoria, BC.

Brothers, T.S., Spingarn, A. 1992. Forest fragmentation and alien plant invasion of central Indiana old-growth forests. Conservation Biology 6: 91-100

Cameron, R. 1996. Wildlife use of cavity tree clumps in clearcuts. Canadian Forest Service – Atlantic, Nova Scotia District Office, R&D Report No. 13, Fredericton, NB.

Campbell, R.W., Dawe, N.K., McTaggart-Cowan, I., Cooper, J.M., Kaiser, G.W., McNall, M.C. 1990. Birds of British Columbia, volume II. Royal British Columbia Museum, Victoria, BC.

Campbell, R.W., Dawe, N.K., McTaggart-Cowan, I., Cooper, J.M., Kaiser, G.W., McNall, M.C., Smith, G.J. 1997. Birds of British Columbia, volume III. Royal British Columbia Museum, Victoria, BC.

Chen, J., Franklin, J.F., Spies, T.A. 1995. Growing-season microclimate gradients from clearcut edges into old-growth Douglas-fir forests. Ecol. Appl. 5: 74-86.

DeGraaf. R.M. 1992. Effects of even-aged management on forest birds at northern hardwood stand interfaces. Forest Ecology and Management 46: 95-110.

DellaSala, D.A., Rabe, D.L. 1987. Response of Least Flycatcher *Empidonax minimus* to forest disturbances. Biological Conservation 41: 291-299.

Dickson, J.D., Conner, R.N., Williamson, J.H., 1983. Snag retention increases bird use of clear-cut. J. Wild. Manage. 47: 799-804.

Eberhart, K.E., and Woodard, P.M. 1987. Distribution of residual vegetation associated with large fires in Alberta. Canadian Journal of Forest Research 17: 1207-1212.

Emlen, J.T., and M. J. DeJong. 1981. The application of song detection threshold distance to census operations. *In* Estimating numbers of terrestrial birds: proceedings of an international symposium held at Asilomar, California. Studies in avian biology No. 6: The American Ornithological Society. *Edited by* C. J. Ralph and M. Scott. Allen Press, Inc., Lawrence. pp. 346-352.

Franklin, J.F. 1993. Preserving biodiversity: species, ecosystems, or landscapes? Ecological Applications 3: 202-205.

Franklin, J.F., Derg, D.R., Thornburgh, D.A., Tappeniner, J.C., 1997. Alternative silvicultural approaches to timber harvests: variable retention harvest systems. In: Kohm, K.A., Franklin, J.F., (Eds.) Creating a forest for the 21st century. Island Press, Washington, DC, pp. 111-140,

Gauthier, J., and Auby, Y. 1996. The breeding birds of Quebec. Canadian Wildlife Service, Montreal, QC.

Godfrey, W.E. 1986. The birds of Canada. National Museums of Canada. Ottawa, ON.

Hansen, A.J., Spies, T.A., Swanson, F.J., and Ohmann, J.L. 1991. Conserving biodiversity in managed forests. BioScience 41: 382-392.

Hansen, A.J., McComb, W.C., Vega, R., Raphael, M.G., and Hunter, M. 1995b. Bird habitat relations in natural and managed forests in the West Cascades of Oregon. Ecol. Applic. 5: 555-569.

Hansen, A.J., Garman, S.L., Weigand, J.F., Urban, D.L., Raphael, M.G. 1995a. Alternative silviculture regimes in the pacific northwest: simulations of ecological and economic effects. Ecol. Appl. 5: 535-554.

Hobson, K.A., and Schieck, J. 1999. Changes in bird communities within boreal mixedwood forest during the first thirty years: contrasting the effects of harvest and wildfire. Ecological Applications in press.

Holt, R.E., Martin, K. 1997. Landscape modification and patch selection: the demography of two secondary cavity nesters colonizing clearcuts. Auk 114: 443-455.

Hunter, M.L. 1993. Natural fire regimes as spatial models for managing boreal forests. Biological Conservation 65: 115-120.

Lambert, F.R., 1992. The consequences of selective logging for Bornean lowland forest birds. Philoe. Trans. R. Soc. Lond. B. Biol. Sci. 335: 443-457.

Laurance, W.F. 1991. Predicting the impacts of edge effects in fragmented habitats. Biological Conservation 55: 77-92.

Lindenmayer, D.B., Franklin, J.F. 1997. Managing stand structure as part of ecologically sustainable forest management in Australian mountain ash forests. Conserv. Biol. 11: 1053-1068.

Lovejoy, T.E., Bierregaard, R.O., Rylands, A.B., Malcolm, R.J., Quintela, C.E., Harper, L.H., Brown, J.R., Powell, A.H., Powell, G.V., Schubart, H.O., and Hays, M.B. 1986. Edge and other effects of isolation on Amazon forest fragments. *In* Conservation biology: the science of scarcity and diversity. *Edited by* M.E. Soule, Sinuaer Associates, Sunderland, MA. pp. 257-285.

MacKenzie, K., and Steventon, D. 1996. Bird use in a patch retention treatment in SBSmc forests. British Columbia Ministry of Forests, Prince Rupert Forest Region, Extension Note 16.

Matthysen, E., Adriaensen, F. 1998. Forest size and isolation have no effect on reproductive success of Eurasian Nuthatches (*Sitta europea*). Auk **115**: 955-963.

Merrill, S.B., Cuthbert, F.J., Oehlert, G. 1998. Residual patches and their contribution to forest-bird diversity on northern Minnesota aspen clearcuts. Conserv. Biol. **12**: 190-199.

Norton, M.R., and Hannon, S.J. 1997. Songbird response to partial-cut logging in the boreal mixedwood forest of Alberta. Canadian J. Forest Resources **27**: 44-53.

SAS Institute Inc. 1989. SAS/STAT User's Guide, Version 6, Fourth Edition, Volume 1 And 2. SAS Institute Inc., Cary, NC.

Saunders, D.A., Hobbs, R.J., and Margules, C.R. 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology **5**: 18-32.

Schieck, J. 1997. Biased detection of bird vocalizations affects comparisons of bird abundance among forested habitats. Condor 99: 179-190.

Schieck, J., Nietfeld, M., and Stelfox, J.B. 1995. Differences in bird species richness and abundance among three successional stages of aspen dominated boreal forests. Canadian Journal of Zoology **73**: 1417-1431.

Semenchuk, G.P. 1992. The atlas of breeding birds of Alberta. Federation of Alberta Naturalists, Edmonton, AB.

Schmiegelow, F.K., Machtans, C.S., and Hannon, S.J. 1997. Are boreal birds resilient to forest fragmentation? An experimental study of short-term community responses. Ecology **78**: 1914-1932.

Seip, D., and Parker, K. 1997. Use of wildlife tree patches by forest birds in the Sub-Boreal Spruce (SBS) zone. British Columbia Ministry of Forests, Prince George Forest Region, Note PG-08.

Stelfox, J.B. 1995. Introduction. *In* Relationships between stand age, stand structure, and biodiversity in aspen mixedwood forests in Alberta. *Edited by* J.B. Stelfox. Alberta Environment Centre, Vegreville, AB. AECV95-R1 pp. 1-12.

Steventon, J.D., MacKenzie, K.L., Mahon, T.E. 1998. Response of small mammals and birds to partial cutting and clearcutting in northwestern British Columbia. Forestry Chronicle **74**: 703-713.

Systat Inc. 1989. SYGRAPH graphics. Evanston, IL.

Taylor, R.J., Haseler, M.E. 1995. Effects of partial logging systems on bird assemblages in Tasmania. For. Ecol. Manage. **72**: 131-149.

ter Braak, C.J. 1992. CANOCO - A FORTRAN program for canonical community ordination. Microcomputer Power, Ithaca, NY.

Weinberg, H.J., Roth, R.R. 1998. Forest area and habitat quality for nesting wood thrushes. Auk **115**: 879-889.

Wolf, A.T., Howe, R.W., and Davis, G.J. 1995. Detectability of forest birds from stationary points in northern Wisconsin. *In* Monitoring bird populations by point counts. *Edited by* C.J. Ralph, J.R. Sauer, and S. Droege, USDA Forest Service, General Technical Report PSW-GTR-149, Albany, CA. pp. 19-23.

CHAPTER 8. THE USE OF RESIDUAL CLUMPS BY BATS IN THE BOREAL FOREST OF NORTHWESTERN ALBERTA

Lui Marinelli

Introduction

Timber harvesting is restructuring the boreal forest mosaic in Alberta. With the recent introduction of aspen into harvesting plans, forests are being reshaped at an even faster rate. Holling (1995) characterized traditional resource management as controlling the variability of ecosystems so that a steady flow of goods and services could be produced for humans. This traditional view of forest value is limited to the monetary worth of present and future timber resources. Emerging environmental, ecological, and social concerns are forcing a much broader perspective of resource-use strategies. Strategies of ecosystem management are being developed (e.g., Alberta Forest Conservation Strategy Steering Committee 1997), encouraging private companies to reassess the value of the forest and modify harvesting practices (e.g., Alberta Pacific Forest Industries Inc. 1996).

Harvesting practices are evolving to include leaving standing trees and snags in harvested areas in an effort to retain pre-harvesting species of flora and fauna. Such forest structure may make harvested areas more desirable to native bats that use the forest environment to forage, roost, and breed (Thomas 1988b). Currently, the pattern in which residual trees and snags are left on the landscape is subjective (Alberta Pacific Forest Industries Inc. 1996). Objective decision-making requires a greater understanding of the effect on biotic communities of retaining residual elements and the manner of their dispersion.

Retaining bats in forest clumps will depend upon requirements of different bat species and sexes. The boreal forest of Alberta supports five species of bats (van Zyll de Jong 1985). These bats show fidelity to roost sites in the forest when they return from hibernaculum or migration. During the spring and summer, cavity-roosting colonial species use a wide variety of cavities including spaces behind exfoliated bark and natural or excavated cavities in large, old snags (Crampton and Barclay 1998, Thomas 1988a,b). Within the mid-boreal forest of Alberta, old forests were considered critical for bats because they contained large, decayed snags, which provide crevices for cavity-roosting bat species (Crampton and Barclay 1998). Hoary bats (*Lasirus cineserus*) generally do not exploit cavities but roost in fixed or shifting locations in foliage of large, live trees (Barclay 1984).

Roosting strategies can be sex-specific (Barclay 1991). Males do not maintain high body temperatures during the day and may select cool roosts to minimize their metabolic rates and diurnal energy expenditures (Thomas 1988b). Females have a reproductive constraint on their ability to exploit the energy saving of torpor because fetal growth rates and presumably milk production are temperature-dependent processes (Racey 1982). During pregnancy and lactation, females maintain body temperatures well above ambient during the day to maximize fetal growth rates (Kunz 1982b, Racey 1982) and as such tend to select warm roost sites.

In this study, harvested stands that varied in the size of residual tree clumps (i.e., number of trees per clump) in two areas were established to evaluate how the flora and fauna would respond to different dispersion patterns of residual trees. The current project will concentrate on differences that are detected in the first two years following harvest, however, throughout the forest rotation the opportunity exists to evaluate the value to bats of the clumps that are created in the experiment.

Objectives

I will compare bat communities among three different harvesting treatments to evaluate which pattern of dispersion of residual trees maximizes bat diversity and abundance. Within each treatment, I also compared bat communities that use the middle and edge of clumps as well as the cutover areas. Pre-harvest sampling was done to assess whether bat communities were similar among treatments prior to harvest. This report will outline the bat response to the treatments in the first year post-harvest. The results from the 1998 field season will be supplemented by data collected in 1999.

Methods

A description of the study areas and experimental harvesting is presented in the Study Area and Methods sections of Chapter 2. Bat detectors were used to estimate species richness and abundance in each treatment in both Whitecourt and Grande Prairie study areas in June and July 1998. Ultrasonic detectors do not provide data that can be translated directly into estimates of population density (Thomas 1988a); they provide a relatively unbiased index of the levels of use or animal abundance among habitats (Thomas 1988b). Detectors consisted of an Anabat IV ultrasonic bat detector (Titley Electronics, AUS), a delay switch to turn the detector on at dusk and off at dawn, a cassette recorder, and a 12-volt battery, all of which were housed in a 1.5 litre (1.6 gal.) Rubbermaid® container. The battery was used to run the detector and the delay switch. A 2-cm (0.8 in.) hole was cut into the side of the container to accommodate the microphone. The sensitivity setting on the detectors was set between 6 and 7 to minimize insect noise. The division ratio on detectors was set to 16 to reduce the frequency of each bat call from the ultrasonic to the audible range so that it could be recorded on the cassette recorder.

Within each stand, three bat detectors were used simultaneously over three days (i.e., nine bat-sampling nights/stand). On each day, a detector was positioned in the middle of a clump, another at the clump edge, and a third within the cutover area. Each morning, detectors were moved to a different clump middle, clump edge and position within the cutover area. After three days, nine different locations were sampled, three in the middle, three at edges, and three in cutover areas.

Detectors were attached 1.5 m (5 ft.) above the ground to trees or metal posts and oriented such that the microphone were pointing 30-degrees above horizontal. Clumps and positions in the cutover area were randomly selected. Bat detectors that sampled the middle of clumps were positioned approximately 20 m (22 yd.) from the centre, with the microphone directed towards the middle. Detectors were affixed to trees that had the most unobstructive line-of-sight to the middle of the clump. Edges were sampled by positioning the detectors on trees such that microphones were parallel with the edge, thus sampling bats on the edge and approximately five metres on either side. Detectors in the cutover area were attached to metal posts that were positioned at least 50 m (55 yd.) from any clump or forest edge.

Two stands in each study area were randomly selected and sampled simultaneously. Detectors were run for three consecutive nights in each stand and were checked each morning to ensure proper functioning. After three days, detectors were moved to new stands.

Bats were identified by analyzing their search-phase echolocation calls. Each sequence of one or more echolocation pulses with <1 s between sequential pulses was recorded as a pass by a bat (Fenton 1970). Calls were categorized based on their average lower frequency into high (≥ 40 MHz) and low (< 40 MHz). Of the bats found in the forest of Alberta, little brown (*Myotis lucifugus*) and northern long-eared (*M. septentrionalis*) bats produce calls in the high range. The hoary, big brown (*Eptesicus fuscus*), and the silver-haired (*Lasionycteris noctivagans*) bats produce calls in the low frequency range. Echolocations were analyzed using Anabat and Analook software (Tilley Electronics, AUS).

The flying insect community was sampled in the middle and at the edge of clumps and in the cutover areas using sticky, flight-intercept traps. Traps consisted of laminated, white, card stock sheets (8 x 11 in.) that were coated with Tangle-Trap (The Tanglefoot Company, MI) and suspended 1.5 m (5 ft.) off the ground. When sampling in the middle and edge of clumps, traps were suspended from a piece of wood that was attached to and extended from a tree. In the cutover areas, traps were attached to the rebar used to hold bat detectors. The insect community was sampled in a stand simultaneous with the sampling of the bat community.

A single trap was setup at each of the nine bat-detector locations, and sampled the flying insect community for three consecutive days. After three days, traps were wrapped in plastic and frozen for later analyses. During the analyses, each side of the trap was divided into four equal quadrates. I randomly selected two quadrates per side to enumerate insects.

Flying insects were enumerated in two size categories: < or ≥ 5 mm (0.2 in.) in length. Those insects greater than 5 mm (0.2 in.)are more likely to be a potential food source for bats (Anthony and Kunz 1977) and were further identified to one of eight orders: Diptera, Hymenoptera, Lepidoptera, Hemioptera, Homoptera, Coleoptera, Neuroptera, and Araneae. The numbers of Diptera and insects less than 5 mm (0.2 in.) that we counted were at least an order of magnitude greater than the remaining seven orders of insects. The orders Neuroptera and Araneae were not considered in the analyses because they occurred in very low numbers.

The method used to sample insects has inherent biases. Large individuals of the Order Lepidoptera and Coleoptera are more likely to escape the traps and therefore be underrepresented in the samples. Colour can have Order-specific effects on the attractiveness of the trap, primarily for those species active during the day. Additionally, sampling at different heights from the ground can produce different insect communities. During this study, flight-intercept traps were all the same colour and position the same height from the ground. As such, the biases inherent with the procedure are similar in all locations.

The effects on bat abundance of all combinations of three tree dispersion patterns (i.e., treatments) and three locations within a stand were compared using ANOVA. The effects on insect abundance were compared using MANOVA. The project was designed to minimize the effect of variation associated with stand groups by statistically blocking for this effect (i.e., randomized block design). The abundance values for bats and insects were log transformed to give less weight to the larger abundance values. In all analyses, results were considered statistically significant if the probability of them occurring by chance was less than 0.05.

Results

Due to difficulties with the bat detection equipment, the study sites in both Whitecourt and Grande Prairie were not sampled for bats to the extent proposed. Incomplete sampling resulted in a reduction in the power of tests to detect differences.

Whitecourt Study Site - Based on 34 bat-detection nights, 102 bat calls were recorded post-harvest (Fig. 8.1b). Of those calls, 68% were categorized as high frequency and 32% as low frequency. Although the sampling effort was less during the 1998 field season, we detected significantly more bat calls than prior to harvesting ($\chi^2 = 24.3$, df = 1, $P < 0.01$). During 108 bat-detection nights prior to the harvesting of the stands (i.e., 1997 field season), fewer calls were detected each night and significantly fewer of them were categorized as low frequency.

Post-harvest the ANOVA model which included treatment, location, and stand group as predictors of bat abundance was non-significant ($F = 1.33$, df = 7, 33, $P = 0.20$), suggesting that the variables used to explain variation in abundance did not account for a significant amount of variation. Because of the low sample size, the likelihood of detecting a difference was relatively poor (Power = 0.52 with $\alpha = 0.05$). More bat calls were detected however in medium size than in large or small clumps (Table 8.1). Additionally more calls tended to be recorded in the cutover areas that surrounded the clumps than at the edge or in the middle of the clumps. The four stand groups in each treatment were relatively similar, accounting for only 8.3 % of the variation in bat abundance.

The insect communities did not differ significantly among treatments (Wilk's Lambda, $F = 0.83$, df = 18, 180, $P = 0.66$), although stands with large clumps tended to have the greatest abundance of insects and medium clumps the least (Table 8.2). Location accounted for a significant amount of the

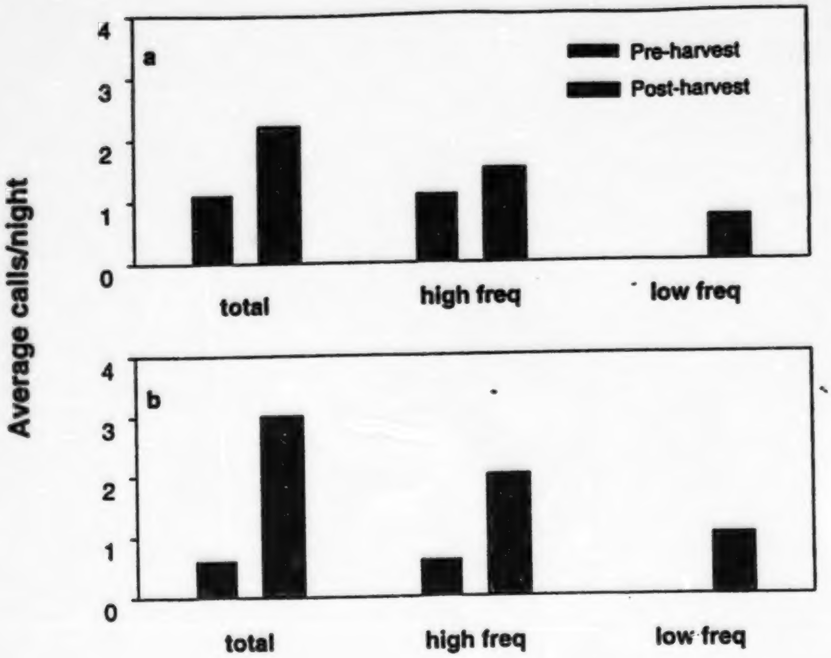


FIGURE 8.1. Average number of bat calls detected each night in a) Grande Prairie and b) Whitecourt study areas during the spring and summer of 1997 (Pre-harvest) and 1998 (Post-harvest). Calls were separated into high and low frequency categories.

TABLE 8.1. Summary statistics for bat abundance in the Grande Prairie and Whitecourt study sites, 1998. The statistics (i.e., mean number of calls per night, standard error [SE], and number of survey nights [N]) are summarized by treatment, stand group, and location.

		GRANDE PRAIRIE			WHITECOURT		
		MEAN	SE	N	MEAN	SE	N
TREATMENT	1	2.7	1.8	13	0.5	0.3	8
	2	1.8	0.6	6	6.6	3.8	14
	3	2.0	0.7	20	0.6	0.4	12
STAND GROUP	A	2.2	0.4	8	1.4	0.2	7
	B	1.6	0.8	17	7.4	6.3	7
	C	1.3	0.7	7	0.4	0.3	12
	D	4.7	3.3	7	5.4	4.0	8
LOCATION	Middle	0.9	0.5	11	0.2	0.2	9
	Edge	3.2	1.0	12	3.2	2.7	12
	Cut-over	2.4	1.5	16	4.8	3.4	13

TABLE 8.2. Summary statistics for insect abundance in the Grande Prairie and Whitecourt study sites, 1998. The statistics (i.e., mean number of calls per night, standard error [SE], and number of survey nights [N]) are summarized by treatment, stand group, and location.

		GRANDE PRAIRIE			WHITECOURT		
		MEAN	SE	N	MEAN	SE	N
TREATMENT	1	156.2	14.1	36	134.0	19.2	35
	2	134.9	9.1	36	101.6	8.3	36
	3	152.7	9.1	36	117.2	10.6	35
STAND GROUP	A	145.2	9.8	27	114.0	22.0	27
	B	198.9	9.2	27	88.9	9.2	26
	C	129.9	16.2	27	120.5	10.9	26
	D	117.7	8.9	27	145.4	15.2	27
LOCATION	Middle	135.5	11.4	36	123.3	17.9	35
	Edge	151.2	11.6	36	140.5	12.9	35
	Cut-over	157.1	10.0	36	89.3	6.3	36

variation in insect communities (Wilk's Lambda, F = 2.62, df = 18 180, P < 0.01). More insects were caught at the edge than in the middle of clumps, or in the cutover area. Stand group D had the greatest abundance of insects, with stand group B having the least; overall stand group accounted for a significant amount of variation in insect community (Wilk's Lambda, F = 2.78, df = 27 258, P < 0.01).

Grande Prairie Study Site - Based on 39 bat-detection nights, 89 bat calls were recorded in the Grande Prairie study area post-harvest (Fig. 8.1a). Of those calls, 69% were categorized as high frequency and 31% as low frequency. Although the sampling effort was less during the 1998 field season, we detected significantly more bat calls than prior to harvest ($\chi^2 = 37.3$, df = 1, P < 0.01). During 108 bat-detection nights prior to the harvest (i.e., 1997 field season), fewer calls were detected each night and significantly fewer of them were categorized as low frequency.

An ANOVA model that included treatments, location, and stand groups as predictors of bat abundance was non-significant (F = 1.63, df = 7 38, P = 0.17), suggesting that the variables used to explain variation did not account, overall, for a significant amount of variation. Because of the low sample size, the likelihood of detecting a difference was relatively poor (Power = 0.57). Large clumps and edges of clumps tended to have the greatest abundance of bat calls (Table 8.1).

The insect community in the Grande Prairie study site did not differ significantly among treatments (Wilk's Lambda, F = 1.63, df = 18, 176, P = 0.06), although stands with large and small clumps tended to have greater insect abundance than did medium clumps (Table 8.2). Insects were more abundance in cutover areas and at the edge of clumps than in the middle of clumps (Wilk's Lambda, F = 2.28, df = 18, 176, P < 0.01). Stand groups accounted for a significant amount of variation (Wilk's Lambda, F = 2.78, df = 27, 258, P < 0.01) with stand group B having the greatest abundance and stand group D, the fewest insects.

Discussion

The new environment produced by harvesting had an affect on the bat community. Five species of bats use the boreal forest of Alberta to breed, roost, and forage (Van Zyll de Jong 1985), each occupying different niches based on where they fly, where they forage, and on what they eat. Prior to the harvesting of trees for this study, the mature forest had a closed canopy with few gaps, presumably providing suitable habitat for under-canopy (e.g., little brown) and above-canopy (big brown) bats. Following harvesting, the new environment had remnants of the forest, but was dominated by cutover areas, a habitat more suited to open-habitat bats (e.g., silver-haired and hoary). The results from 1998 suggest that the bat community post-disturbance is different from that pre-disturbance. In the cutblocks, more bats calls were enumerated with many of these being low-frequency calls (LF). These results may be due to a change in species diversity, animal abundance, or both following harvest.

Given the study's preliminary results, the pattern in which residual trees were left on the landscape appears to be inconsequential to bats. In both study areas, bat abundance did not differ statistically among the treatments, nor were there obvious patterns between the study areas.

Although results were not statistically significant more bats tended to be detected in cutover areas and at the edge of clumps than in the middle of clumps. This result was similar in both study areas.

Because of technical problems, we were unable to sample the treatments and locations sufficiently during 1998, resulting in low power to detect differences if they exist. As such, attempting to describe potential relationships between bat abundance and treatment or locations within a stand is premature. Relationships, if they exist, should be clearer with additional data from 1999.

Bats of the boreal forest are insectivores (Van Zyll de Jong 1985), feeding on a wide variety of insects (Van Zyll de Jong 1985, Anthony and Kunz 1977). In both study areas, insects were of similar abundance among treatments. Location within the stand was an important predictor of insect abundance, but the patterns were different between study areas and different from the trends in bat abundance. This result is not surprising as the bat and insect communities considered here are made up of species with very different life histories. More promising results may be available with a greater sample size (i.e., greater power), and with finer scale analyses (i.e., LF versus HF bats, species analysis).

Future Research

Bat communities will be surveyed again 1999 to supplement the data collected in 1998, in an effort to evaluate the effect that the different harvesting methods have on bats. The methodology for the detection of bat calls in 1999 will be the same as that done in 1998. A number of the difficulties that lead to incomplete sampling in 1998 have been addressed (i.e., electronic malfunctions with the bat recorders and poor batteries) for the next field season. In addition efforts will be made to better identify the bat community (i.e., mist netting and recording calls directly to the computer)

Literature Cited

Alberta Forest Conservation Strategy Steering Committee. 1997. Alberta Forest Conservation Strategy. A New Perspective on Sustaining Alberta's Forests. Alberta Lands and Forest Service, Alberta Environmental Protection, Edmonton, Alberta.

Alberta-Pacific Forest Industries Inc. 1996. An operator's guide to stand structure. Alberta-Pacific Forest Industries Inc., Boyle, AB

Anthony, E.L.P. and Kunz, T.H. 1977. Feeding strategies of the little brown bat, Myotis lucifugus, in southern New Hampshire. Ecology **58**: 775-786.

Barclay, R.M.R. 1991. Population structure of temperate zone insectivorous bats in relation to foraging behaviour and energy demand. J. Anim. Ecology **60**: 165-178.

Barclay, R.M.R. 1984. Observations on the migration and behaviour of bats at Delta Marsh, Manitoba. Can. Field-Nat. **98**: 331-336.

Crampton L.H. and Barclay R.M.R. 1998. Selection of roosting and foraging habitat by bats in different-aged aspen mixedwood stands. Conser. Biol. **12**: 1347-1358.

Fenton, M.B. 1970. A technique for monitoring bat activity with results obtained from different environments in southern Ontario. Can. J. Zool. **48**: 847-851.

Holling, C.S. 1995. What barriers? What bridges? *In* Barriers and bridges to the renewal of ecosystems and institutions. *Edited by* Gunderson, L.H., Holling, C.S., and Light S.S. Columbia University Press, New York. pp 3-34

Kunz, T.H. 1982. Roosting ecology. *In* Ecology of Bats. *Edited by* T.H. Kunz. Plenum Press, New York. pp 1-55.

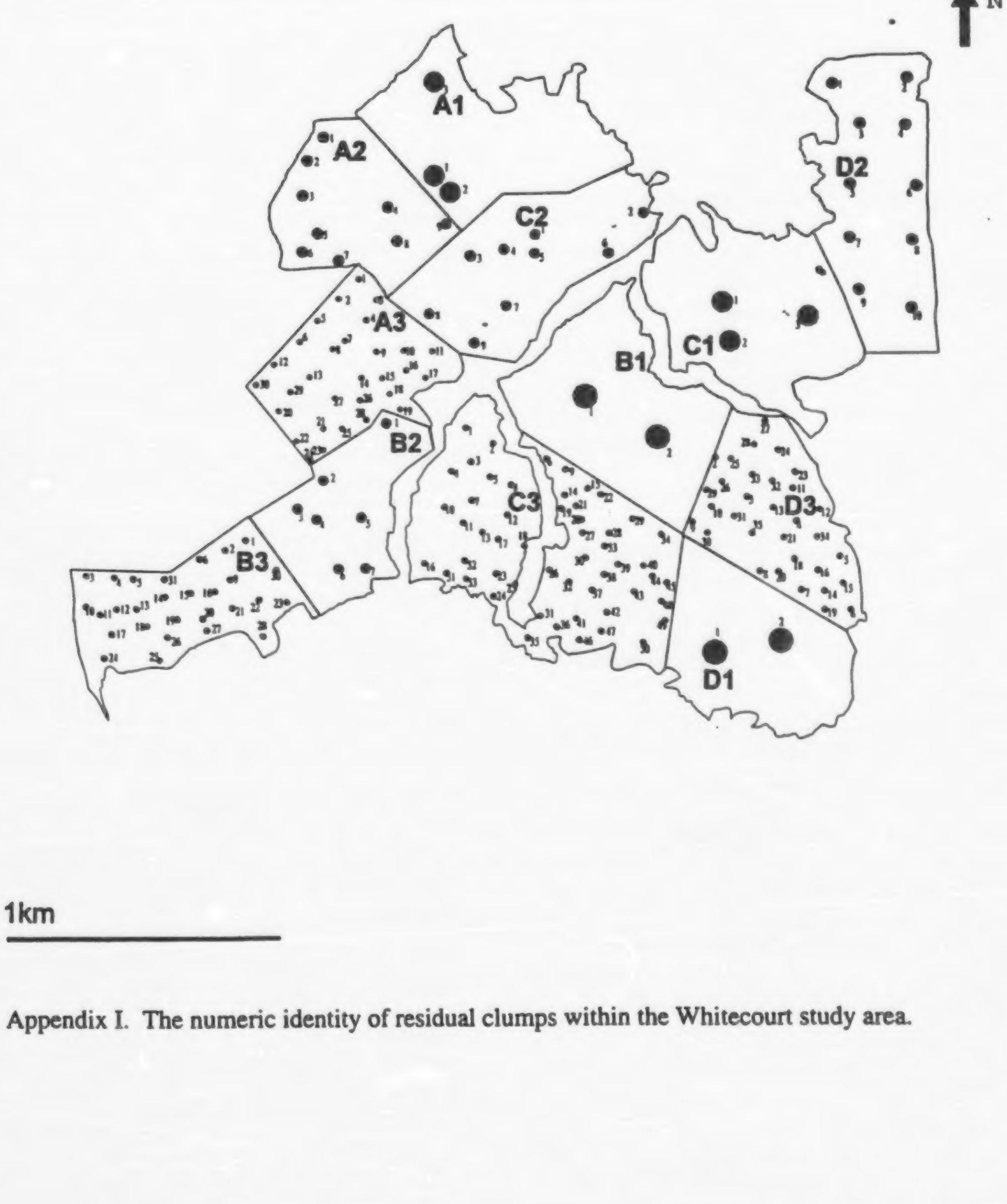
Thomas, D.W. 1988a Census and survey techniques. *In* Behavioural and ecological methods for the study of bats. *Edited by* T.H. Kunz. Smithsonian Institute Press, Washington, D.C. pp 77-89

Thomas, D.W. 1988b. The distribution of bats in different ages of douglas-fir forests. J. Wildl. Manage. **52**: 619-626.

Racey, P.A. 1982. Ecology of bat reproduction. *In* Ecology of Bats. *Edited by* T.H. Kunz. Plenum Press, New York. pp 57-104

Van Zyll de Jong, C.G. 1985. Handbook of Canadian Mammals: Bats. Natural Museums of Canada. Ottawa.

APPENDIX I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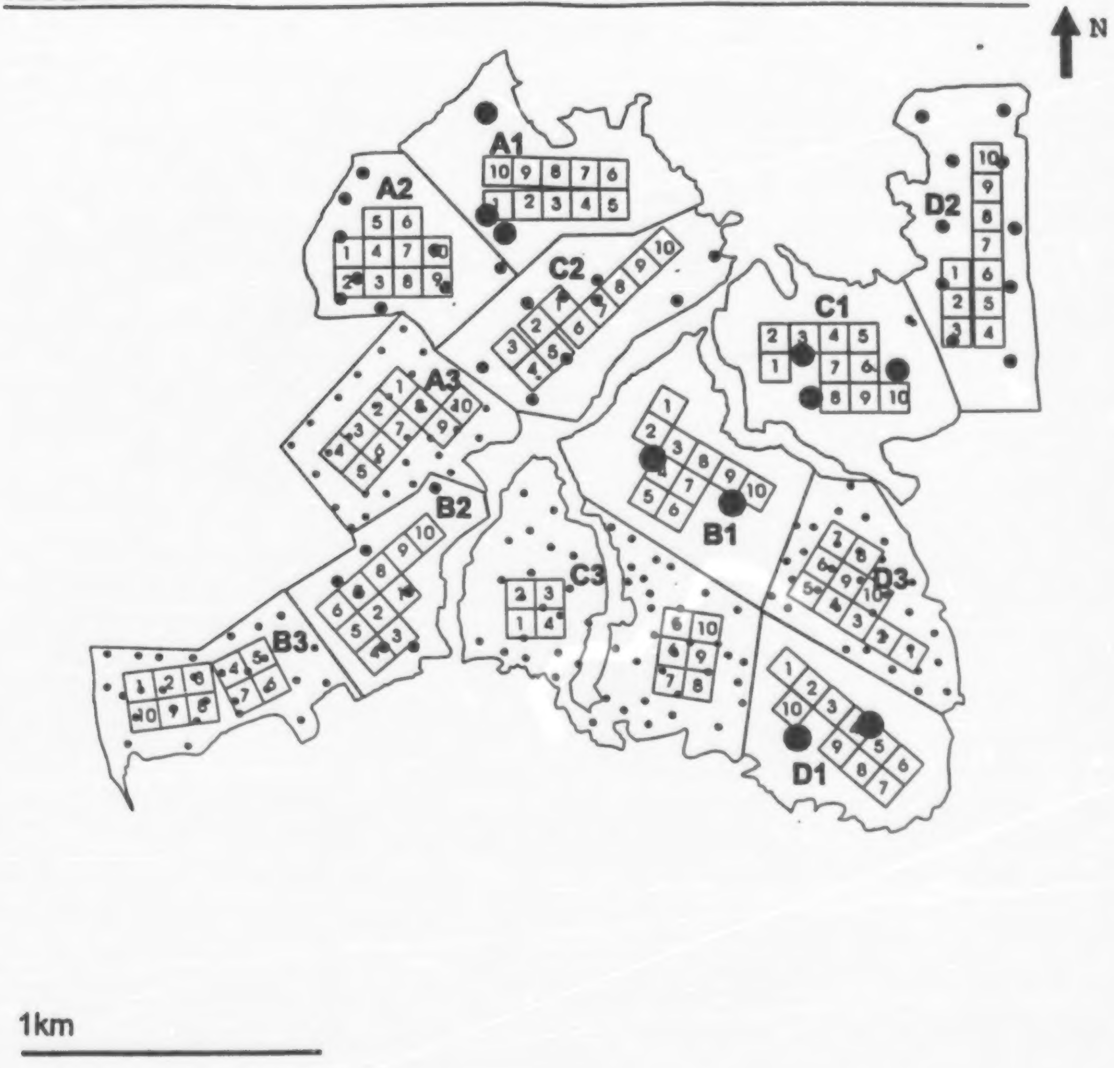
D2
D3
1km

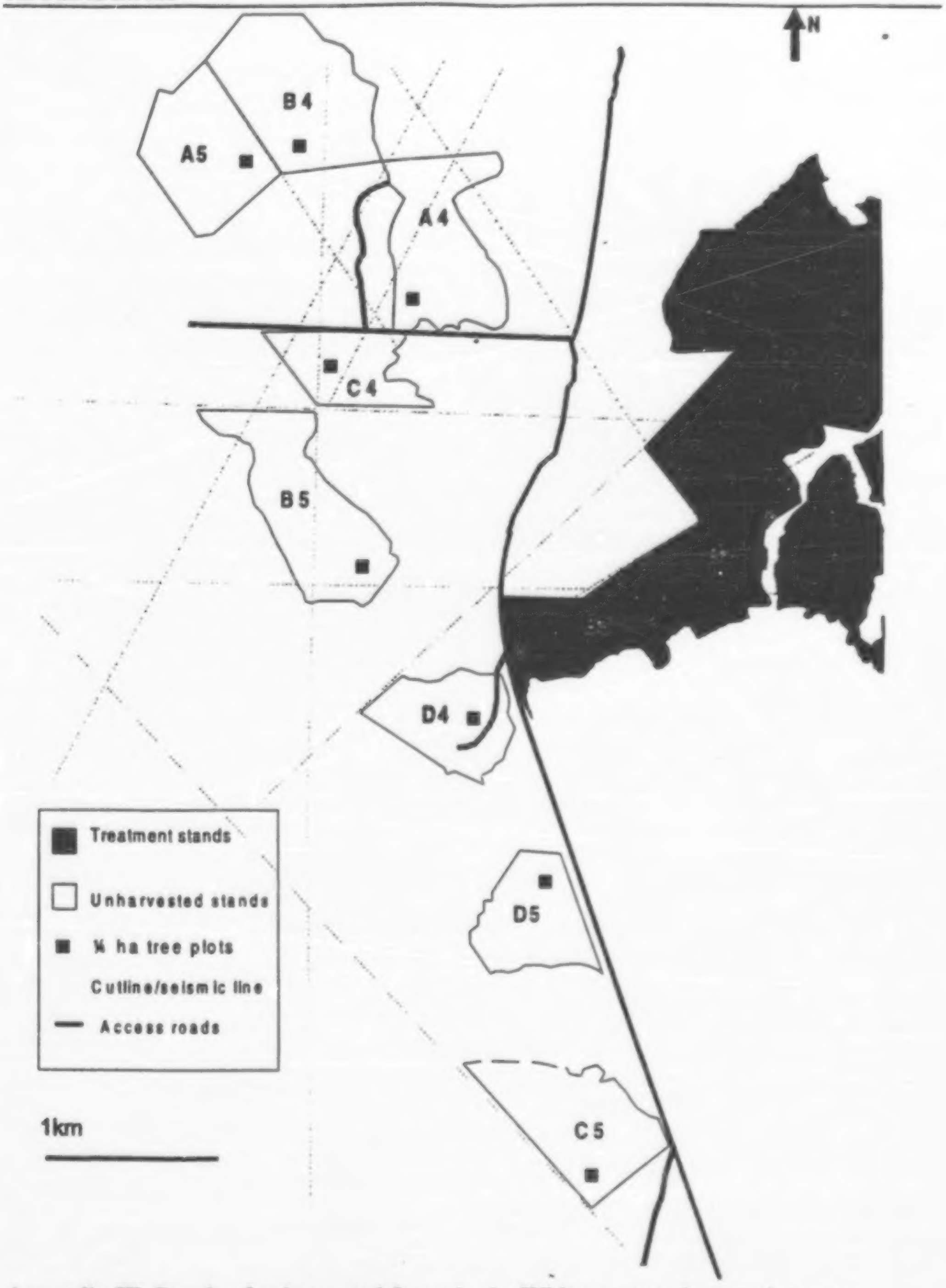
Appendix I. The numeric identity of residual clumps within the Whitecourt study area.

APPENDIX II


N
A1
A2
A3
B1
B2
B3
C1
C2
C3
D1
D2
D3
1km

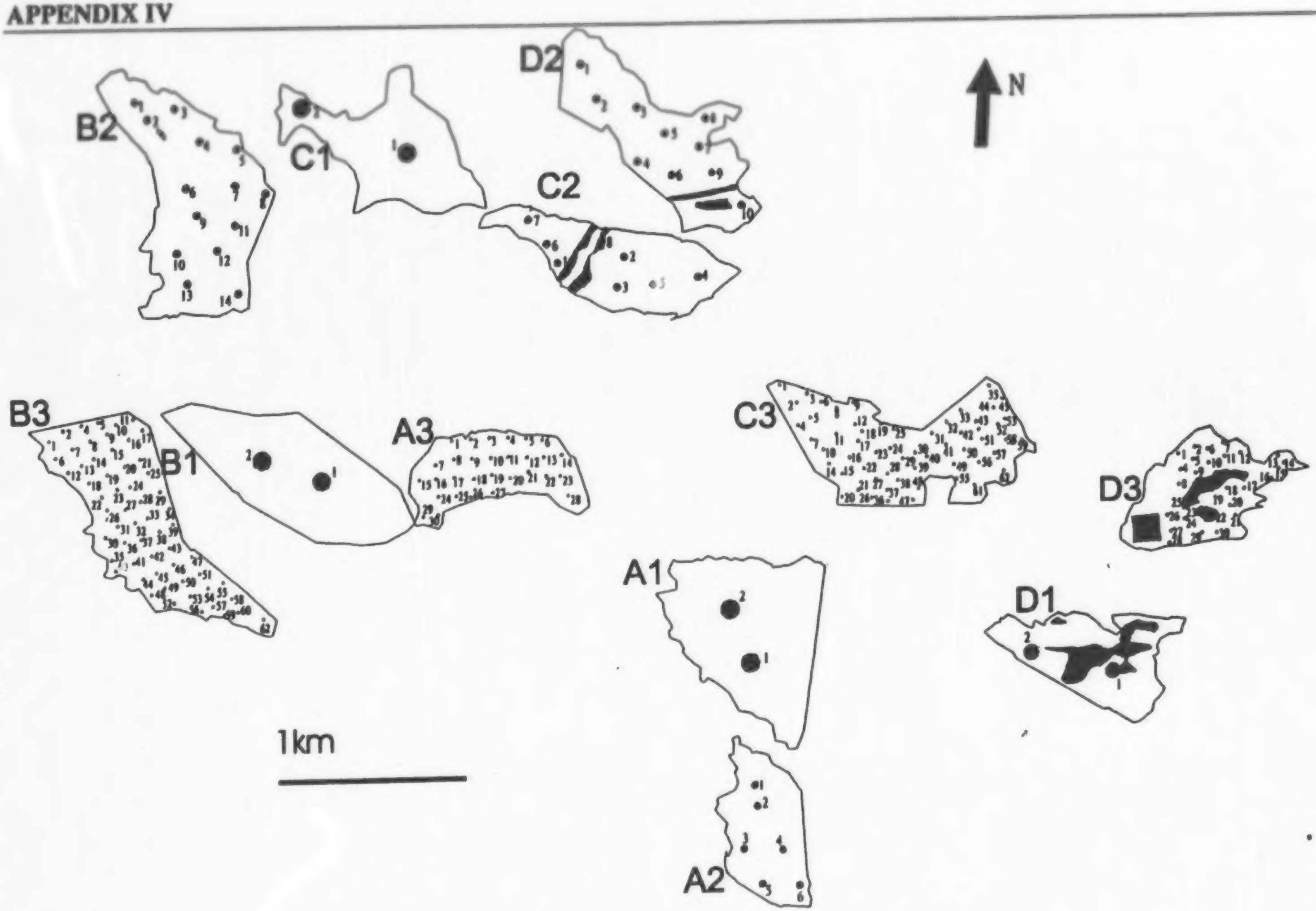
Appendix II. Whitecourt study area showing the location of the ten 1-ha (2.47 acre) sites within each stand.

APPENDIX III


N
B 4
A5
A 4
C 4
B 5
D4
D5
C 5
Treatment stands
Unharvested stands
¼ ha tree plots
Cutline/seismic line
Access roads
1km

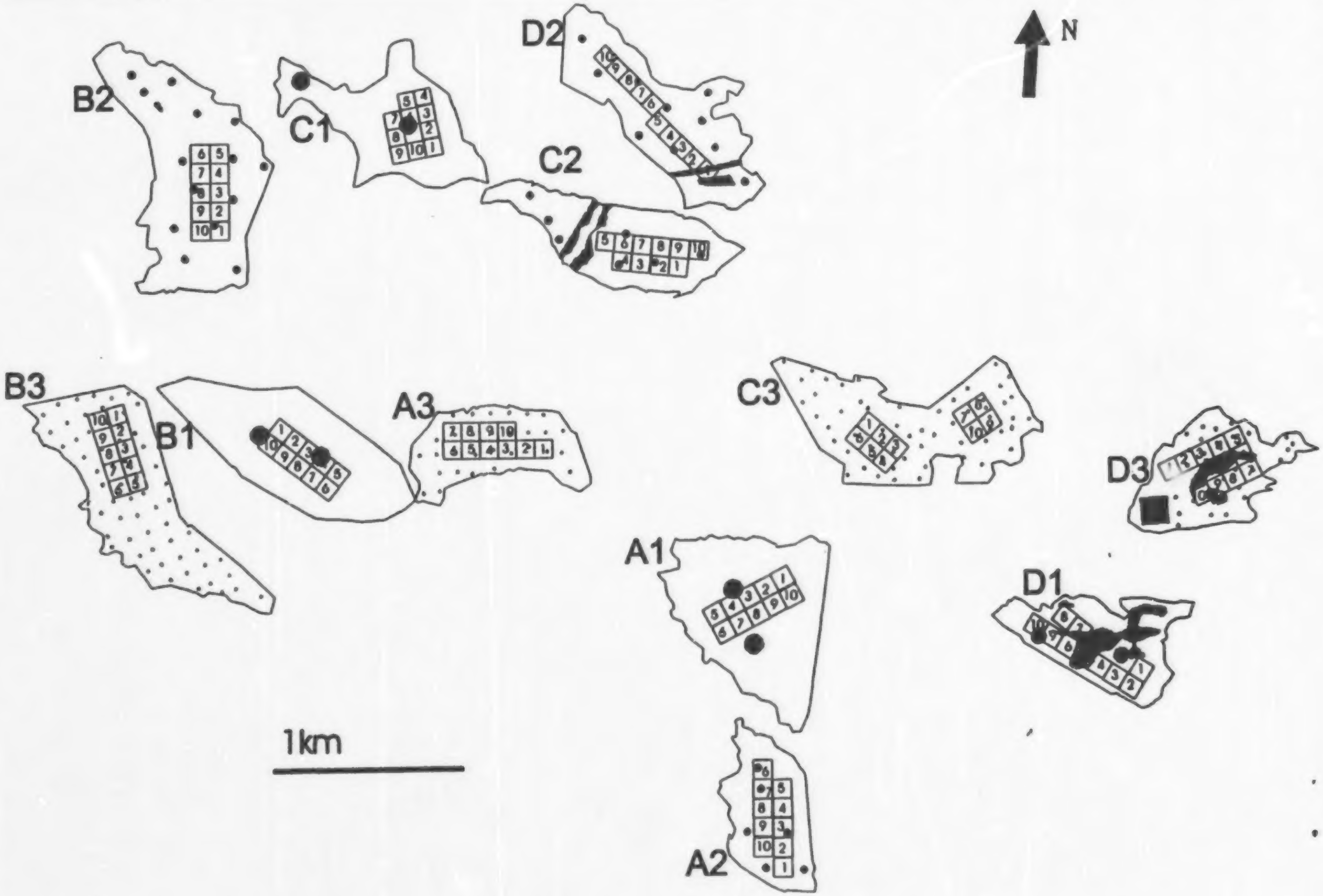
Appendix III. Stands of unharvested forest in the Whitecourt study area that were surveyed for falldown of trees. ¼ ha tree plots are not to scale, second pass stands (ie. 4's) also had 150 m (164 yd.) beetle transects.

APPENDIX IV


B2
C1
D2
C2
N
B3
B1
A3
C3
D3
A1
D1
A2
1km

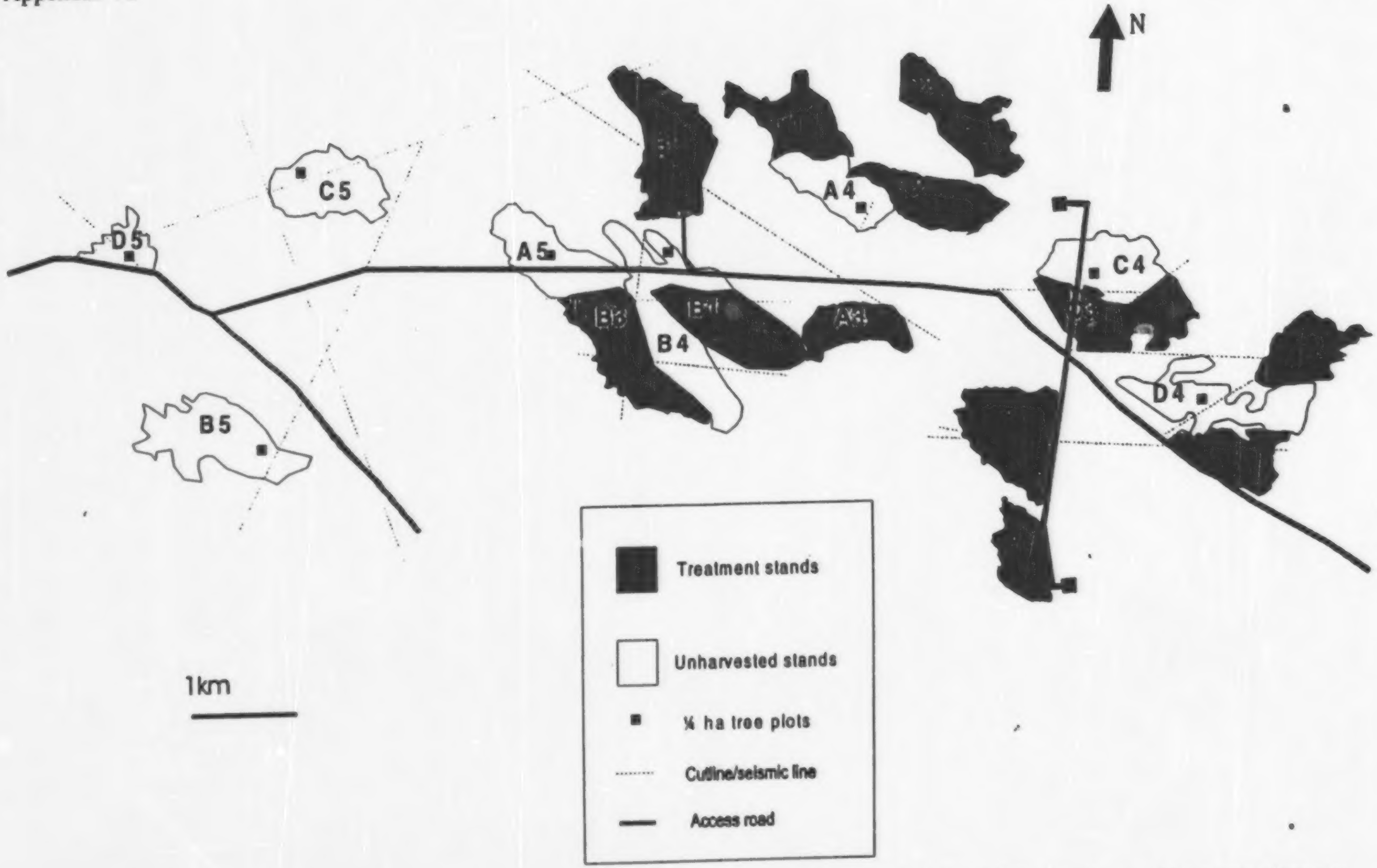
Appendix IV. The numeric identity of residual clumps within the Grande Prairie study area.

APPENDIX V


N
B2
C1
D2
C2
B3
B1
A3
C3
D3
A1
D1
1km
A2

Appendix V. Grande Prairie study area showing the location of the ten 1-ha (2.47 acre) sites within each stand.

Appendix VI


N
C5
D5
B5
A5
A4
B3
B1
B4
A3
C4
D3
D4
1km
Treatment stands
Unharvested stands
¼ ha tree plots
Cutline/seismic line
Access road

Appendix VI. Stands of unharvested forest in Grande Prairie study area that were surveyed for falldown of trees. ¼ ha tree plots are not to scale; second pass stands (i.e. 4's) also had 150 m (164 yd.) beetle transects.